MOUNTAINTOP REMOVAL MINING/VALLEY FILL ENVIRONMENTAL IMPACT STATEMENT TECHNICAL STUDY

PROJECT REPORT FOR TERRESTRIAL STUDIES

Terrestrial Vertebrate (Breeding Songbird, Raptor, Small Mammal, Herpetofaunal) Populations of Forested and Reclaimed Sites

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Executive Summary

In this study, we quantified diversity and relative abundance of songbird, raptor, small mammal, and herpetofaunal populations on 4 treatments: 2 ages of reclaimed mountain top mining/valley fill (MTMVF) areas (younger grassland; older shrub/pole-size), fragmented forests predominantly surrounded by reclaimed land, and large tracts of intact forest. Our first objective was to quantify the richness and abundance of the wildlife community in relatively intact forest sites of the pre-mining landscape and in the grassland, shrub/pole, and fragmented forest sites of the post-mining landscape to provide objective data on gains and losses in terrestrial wildlife communities. Specifically for species that require forested habitats, we compared abundance of species in intact and fragmented forests. Our second objective was to quantify nesting success of grassland birds on the reclaimed grassland sites because grassland birds are declining in the U.S. partially due to loss of habitat, and some have suggested that these newly created grasslands are providing important habitat for grassland species.

Songbirds

For songbirds, overall richness and abundance were highest in the shrub/pole treatment, which was not surprising since the mix of habitat conditions provides more niches for greater bird diversity. These shrub/pole habitats were dominated by edge species. The grassland treatment had lowest richness and abundance, again not too surprising since grassland bird communities tend to be the least diverse. The bird community in the grassland habitat was dominated by birds in the grassland guild; though edge species were fairly common because of shrub plantings in some areas. We found no statistical difference in overall bird richness and abundance between intact and fragmented forests because increased abundance of edge and interior-edge species in fragmented forests balanced the loss of forest-interior species. Forest-interior species were significantly more abundant in the intact forest. Forest-interior species are affected 2 ways by mountaintop mining; first by a reduction in the total amount of forested habitat available and second by decreased abundance in the remaining fragmented forest.

Generally, the bird community shifted from predominantly forest interior species in the intact forests to edge and grassland species in the reclaimed areas.

Because some songbird species are known to respond negatively to forest fragmentation, we examined abundances of individual species in intact and fragmented forests. The Acadian Flycatcher, American Redstart, Hooded Warbler, Ovenbird, and Scarlet Tanager had significantly higher abundances in intact forests during at least one year of the study, suggesting that fragmentation of the landscape is having on effect on abundance of these species. Distance from mine/forest edge was a significant predictor for presence of Acadian Flycatchers, Black-and-white Warblers, Yellow-throated Vireos and Scarlet Tanagers. However, Red-eyed Vireos, Indigo Buntings, American Goldfinch, Downy Woodpeckers, Northern Parulas, Pileated Woodpeckers, and Yellow-billed Cuckoos had greater abundances in fragmented forests in at least 1 year of the study. However, because of the large size of most MTMVF areas, it is possible that they may have severe negative effects on populations of forest interior species that require large blocks of unfragmented forest for breeding. The severity of the habitat loss/fragmentation also will depend on whether or not MTMVF areas are re-forested or if they remain in early stages of succession. Non-timber post-mining land uses such as grazing or development will result in permanent fragmentation of forest habitats

Eight grassland bird species were detected in the grassland treatment. Grasshopper Sparrows were the most abundant species, and Eastern Meadowlarks were second most abundant. Henslow's Sparrows and Vesper Sparrows were rare on our sites. Vegetation characteristics were not particularly suitable for them. Bobolinks were rare and did not appear to be breeding on the study sites. We found evidence of breeding for both Dickcissels and Horned Larks. The Savannah Sparrow is fairly common on other grassland sites in West Virginia, but it was absent from our study sites.

We conducted nest searching and monitoring in grassland habitats and focused our efforts on Grasshopper Sparrows, the most common species. Our study sites had low nest densities for this species (0.06 nests/ha), and 36% of nests monitored successfully fledged young. A study in northern West Virginia on reclaimed contour mines found 0.11 nests/ha with 7% nest success. Other grasslands in 4 studies throughout the eastern and midwestern U.S. had 0.06-0.25 nests/ha with 11-41% nest success. Nest densities seemed low on our study sites based on the high number of singing males that were detected during point counts and compared to other studies. Nesting success, however, was at the upper end of the range found in other studies. Because nest densities were so low, we could not determine if grassland habitats on reclaimed mountaintop mine sites are able to sustain viable populations of grassland bird species.

In summary, MTMVF areas provided breeding habitat for both grassland and early successional species. Grassland, edge, and interior-edge songbirds were more abundant on the post-mining landscape. The highest bird species richness was found in the shrub/pole treatment and the lowest was found in the grassland treatment. Richness in fragmented forest and intact forest fell between these 2 treatments. Ponds on MTMVF areas also provided habitat for waterfowl, wading birds, swallows, and shorebirds, primarily during migration. No federally-listed endangered or threatened species were detected, but 3 grassland species (Grasshopper Sparrow, Henslow's Sparrow, and Bobolink) considered rare in West Virginia were observed. . However, abundance of the forest interior guild, some forest interior species (e. g. Ovenbird and Acadian Flycatcher) were significantly lower in fragmented forest than in intact forest. Some forest species also were detected more frequently at points further from mine/forest edges. Populations of forest birds will be detrimentally impacted by the loss and fragmentation of mature forest habitat in the mixed mesophytic forest region, which has the highest bird diversity in forested habitats in the eastern United States. Fragmentation-sensitive species such as the Cerulean Warbler, Louisiana Waterthrush, Worm-eating Warbler, Black-and-white Warbler, and Yellow-throated Vireo will likely be negatively impacted as forested habitat is lost and fragmented from MTMVF. Grassland birds nesting on MTMVF areas had nest survival rates similar to those found in the literature, but some species, particularly the Grasshopper Sparrow and Dickcissel, appeared to have high proportions of unmated males in their populations. Further research is necessary to adequately determine the impacts of MTMVF on the nest survival and population dynamics of grassland-nesting bird species.

Raptors

Thirteen species of raptors were observed during the study in 1 or more of the treatments. Of the 6 species typically associated with forested habitats, the Red-shouldered Hawk was the most common. Their abundance was greater in intact than in fragmented forests. Of the 7 species typically associated with more open habitats, the American Kestrel, Northern Harrier, Red-tailed Hawk, and Turkey Vulture were commonly observed as expected. Rough-legged Hawks and Short-eared Owls were observed in low numbers in the grassland treatment. They are more northern species that use large areas of open habitat and are rarely seen in West Virginia. A pair of adult Peregrine Falcons was observed throughout the summer on the Daltex mine in grasslands surrounding a highwall. The falcons often used the highwall for perching, but we found no evidence of breeding. Generally, these results suggest that MTMVF has resulted in a shift from a woodland raptor community to a grassland raptor community.

Small mammals

Species richness of small mammals did not differ between the 4 treatments in either 1999 or 2000. For overall abundance, there was no significant difference between the 3 treatments sampled in summer 1999. In summer 2000, however, we had increased abundance in grassland and shrub/pole treatments and decreased abundance in the 2 forest treatments with a significant difference between these 2 groups. *Peromyscus* spp. (white-footed and deer mice) were by far the most common species and they mirrored this pattern. These yearly differences were quite possibly due to weather patterns. A severe drought and high temperatures in 1999 could have affected small mammal populations in the grassland community more severely. In 2000, the extremely wet and cool conditions probably benefitted animals in the grassland habitat but adversely affected those in forested habitats.

Two other commonly captured species were chipmunks and short-tailed shrews. Both species were significantly more abundant in intact forests. The relationship for shrews holds only for 1999 when this species was common; it was rarely captured in 2000. House mice were captured only in grasslands. A species that we did not expect to find was the Allegheny woodrat. This species has been declining throughout the Northeast and is typically found using rock outcrops in forested habitats. We captured woodrats at 10 of 20 sites trapped. Capture sites were rip-rap drainage channels that had large boulders with a network of openings and some canopy cover. We captured 26 individuals, including males, females and juveniles, which suggests that some of these sites have a breeding population. However, we did not trap extensively at rock outcrops in forested habitats, so we cannot compare abundance of this species between intact forest and reclaimed sites.

Although bats and large mammals are an important part of the mammalian fauna, we did not examine impacts of MTMVF on these species because of logistical and time constraints.

Our study is in agreement with most literature surveyed in that we found small mammals to be more abundant at early stages of succession than in forest. This trend in our study was driven by the white-footed mouse, a species that is often most abundant in early successional stages (e.g. Hansen and Warnock 1978, Buckner and Shure 1985). Two species, short-tailed shrew and eastern chipmunk, were more abundant in intact forest than fragmented forest. Allegheny woodrats were captured at several shrub/pole sites where rock drains with large boulders and some canopy cover provided useable habitat.

Herpetofauna

Although the overall abundance and richness of the herpetofaunal community sampled from March through September 2000 did not differ statistically between our 4 treatments, we observed a shift from a majority of amphibian species in the 2 forested treatments to a majority of reptile species in the grassland and shrub/pole treatments. In particular, salamander species decreased while snake species increased. Summer 2000 had much more rainfall than normal which provided ample breeding habitat for toads and frogs, a group that accounted for a high proportion of species and individuals in all treatments. Thus, we may have found a more pronounced shift during a drier summer. Herpetofaunal species that require loose soil, moist conditions, and woody or leaf litter ground cover generally were absent from reclaimed sites. Minimizing soil compaction, establishing a diverse vegetative cover, and adding coarse woody debris to reclaimed sites would provide habitat for some herpetofaunal species more quickly after mining. In areas disturbed by clearcutting, researchers have found that salamander populations appear to require many years to recover to pre-disturbance levels. MTMVF results in greater soil disturbance than clearcutting so a longer time may be required for recovery of salamander populations in reclaimed mine sites.

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Terrestrial Vertebrate (Breeding Songbird, Raptor, Small Mammal, Herpetofaunal) Populations of Forested and Reclaimed Sites

Background and Justification

Fragmentation and loss of forest habitat from a variety of human-induced disturbances are major issues in wildlife conservation due to negative effects on a number of wildlife species. Because West Virginia is predominantly forested, it provides important habitat for a variety of terrestrial wildlife species that require large tracts of unbroken forest. Mountaintop mining/valley fill (MTMVF), one type of human-induced disturbance to habitat, sets back successional stages, essentially converting large areas of mature hardwood forest to early successional habitat. Forested valleys located below the target coal seams and beyond the reach of the valley fills often appear vegetatively similar to nearby contiguous tracts of forest, but are partially surrounded by actively mined or reclaimed areas resulting in large amounts of edge habitat. Forest edges exhibit numerous changes in biotic and abiotic factors that can negatively affect plant and animal communities (reviews by Yahner 1988, Paton 1994, Murcia 1995). Thus, species composition and diversity in a reclaimed landscape (one composed primarily of early successional habitats with forest remnants) is expected to change from that of a primarily forested landscape.

Many species of songbirds have shown significant population declines over the last several decades (Askins et al. 1990, Smith et al. 1992), including forest-interior species that depend on large, unbroken tracts of hardwood forest and others that are dependent on early successional habitats. Smith et al. (1992) and Rosenberg and Wells (1995) have documented that some avian populations in West Virginia are stable or increasing whereas these same species are declining in other parts of the eastern United States. Therefore, West Virginia has been identified as an important area in the eastern United States for maintenance of bird populations, particularly those of forest-interior species (Rosenberg and Wells 1995). Both conversion and fragmentation of forested habitats associated with MTMVF can have negative effects on the abundance, diversity, and reproductive success of forest-interior songbird populations (Finch 1991, Robinson et al. 1995). Simultaneously, this mining technique creates early successional habitats that are important to other groups of songbird species. Consequently, there is a tradeoff between bird populations in mature forests with those in early successional habitats, but the extent of change in species composition and diversity is not well quantified.

Large-scale MTMVF also raises questions concerning impacts on raptor populations. Several raptor species, particularly the Red-shouldered Hawk (scientific names of all bird species mentioned in the text are found in Appendix 1), are considered primarily forest species and breed in large tracts of contiguous, mature forest (Hall 1983, Crocoll 1994). Conversion of forest tracts to earlier successional habitats will change the raptor community in an area from predominantly forest-dependent species to open country species. Creation of fragmented forest patches also may decrease the suitability of forests remaining on or near MTMVF areas and lead to lower abundance of forest raptor populations. Previous studies have examined habitat and perch use by raptors on surface mines other than MTMVF areas (Mindell 1978, Forren 1981). We found no published studies comparing forested sites with reclaimed sites. The fragmentation of forest and creation of edge by MTMVF areas may have variable effects on raptor species. Greater amounts of edge can decrease suitability of an area for Red-shouldered Hawks but increase suitability for Red-tailed Hawks (Moorman and Chapman 1996) and increase competition between these species (Bednarz and Dinsmore 1981, Moorman and Chapman 1996). Species often observed hunting in open areas, such as American Kestrels and Northern Harriers (Bent 1937, Forren 1981), may benefit from open areas created by MTMVF, but low availability of suitable perches in open areas may limit use of reclaimed mine lands (Mindell 1978, Bloom et al. 1993). Thus, it is important to quantify what effect relatively large-scale MTMVF areas are having on raptor abundance, diversity, and habitat use.

Small mammals are an important component of biological diversity, and their populations are affected by forest fragmentation (e.g. Gottfried 1977). Further, small mammals are the primary prey base for a variety of mammalian and avian predators; thus changes in their abundance can affect other species. They make up a significant percentage of the diet of many animals, including hawks (Acciptrinae), owls (Strigidae and Tytonidae), red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoargenteus*), coyote (*Canis latrans*), and weasels (*Mustela* spp.) (Mindell 1978, Yearsley and Samuel 1980, McGowan and Bookout 1986). Additionally, small mammals are an important part of the food web as predators, herbivores, and detritivores, and they act as seed dispersers for many plant species (Mumford and Bramble 1973, Bayne and Hobson 1998).

Although we found no previous studies of small mammal populations on MTMVF areas, there have been several studies of small mammals on strip-mined lands throughout the coal mining regions of the mid-western and eastern United States (Verts 1957, De Capita and Bookout 1975, Sly 1976, Hansen and Warnock 1978, Urbanek and Klimstra 1980, McGowan and Bookout 1986). Several of these studies found that small mammal communities on mines differ as a function of time since mining activity ceased (Verts 1957, Sly 1976, Hansen and Warnock 1978, McGowan and Bookout 1986). Three studies compared small mammal populations on reclaimed lands with those on unmined areas (De Capita and Bookout 1975, Kirkland 1976, Urbanek and Klimstra 1980). However, results from these studies differed, with diversity and abundance greater on unmined lands in 1 study (Kirkland 1976) and on reclaimed land in another (Urbanek and Klimstra 1980). Further, unmined lands in the third study (De Capita and Bookout 1975) included habitats other than intact forests which can confound results. Consequently, additional research is needed to clarify the effects of MTMVF on small mammal populations.

Amphibians are the most abundant vertebrates in many temperate forest ecosystems (Burton and Likens 1975), but declines in their populations have been documented worldwide due to various causes including loss and degradation of habitats (Wyman 1990). Amphibian life-history traits make them especially sensitive to disturbances that alter microhabitat and microclimate characteristics (Feder 1983, Sinsch 1990, Stebbins and Cohen 1995). Thus, herpetofauna, particularly amphibians, can be ideal indicators of how well reclamation efforts have succeeded because they are susceptible to small environmental changes (Jones 1986) and make up a large part of the vertebrate biomass on certain sites (Pais et al. 1988, Heyer et al. 1994). However, a thorough literature search revealed little previous research concerning the effects of surface mining on herpetofauna. Myers and Klimstra (1963) and Fowler et al. (1985) studied the colonization of surface mine sediment ponds by herpetofauna, but we found no published literature regarding the effect of surface mining on stream, riparian, or terrestrial herpetofauna. A study of herpetofauna using ponds on MTMVF areas was recently completed (T. Pauley, personal communication), but these data are not currently available. Because the conditions resulting from MTMVF and subsequent reclamation are dramatically different from those provided by the original intact forest, more information is needed on how hepetofaunal populations are responding to these changes.

In our study, we quantified diversity and relative abundance of songbird, raptor, small mammal, and herpetofaunal populations on 4 treatments: 2 ages of reclaimed MTMVF areas (younger grassland; older shrub/pole-size), fragmented forests predominantly surrounded by reclaimed land, and large tracts of intact forest. Our first objective was to quantify the richness and abundance of the wildlife community in relatively intact forest sites of the pre-mining landscape and in the grassland, shrub/pole, and fragmented forest sites of the post-mining landscape to provide objective data on gains and losses in terrestrial wildlife communities. Specifically for species that require forested habitats, we compared abundance of species in intact and fragmented forests. Our second objective was to quantify nesting success of grassland birds on the reclaimed grassland sites because grassland birds are declining in the U.S. partially due to loss of habitat, and some have suggested that these newly created grasslands are providing important habitat for grassland species.

Review of Current Literature

Songbirds

The effects of surface mining activities on bird populations have been examined more than any other taxonomic group. Many studies were conducted in the late 1970's and early 1980's after areas mined in the late 1960's and early 1970's were either reclaimed or revegetated through natural succession (Yahner 1973, Yahner and Howell 1975, Chapman 1977, Crawford et al. 1978, Whitmore 1978, Whitmore and Hall 1978, Wray et al. 1978, Allaire 1979, Whitmore 1979, Wray 1979, Wackenhut 1980, Whitmore 1980, Strait 1981, LeClerc 1982, Wray 1982, Wray et al 1982). Allaire (1980) conducted a thorough review of ornithological literature pertaining to avian use of surface mines during all seasons.

The effects of surface mines on songbirds can be categorized several ways. First, studies can be examined based on the type of mining activity: area-wide, contour, surface, or mountaintop removal, and Allaire (1980) provides a thorough review of studies by the type of mining activity. Studies also can be separated by the hypotheses being examined. In most cases, studies fall into 1 of 3 types: 1) bird use of mines ; 2) bird-habitat relationships; and 3) reproductive success of songbirds on mines. In this review, we examine studies based on the hypotheses being tested and summarize major findings pertaining to bird use of surface mines during the breeding season, incorporating information from Allaire (1980) on MTMVF.

Avian Use of Reclaimed Mines

Most studies of avian use of small surface mines indicate that birds follow a pattern of use typical of that seen in natural succession. The bird community of recently revegetated areas is composed of grassland bird species, typically dominated by Grasshopper Sparrows, Eastern Meadowlarks, Savannah Sparrows, Vesper Sparrows, Horned Larks, and Red-winged Blackbirds. In addition, several authors have noted that the presence of reclaimed mines in eastern states have allowed the range expansion of several grassland species, including Savannah Sparrows, Dickcissels and Bobolinks (Chapman 1977, Whitmore 1978, Whitmore and Hall 1978, Allaire 1979, LeClerc 1982, Wray 1982).

As succession proceeds on mines, the songbird community also changes. Brewer (1958) was the first to study the use of strip mines by songbird species. He examined bird populations on a naturally revegetated mine in Illinois and found 44 species using the area. Most species were forest-edge species, but species composition changed as succession proceeded towards hardwood forest. Karr (1968) found that bird species diversity increased as succession proceeded on strip mines in Illinois. Typical species noted in the shrub/pole phase of succession included Field Sparrows, Gray Catbirds, Brown Thrashers, Indigo Buntings, Yellow Warblers, Prairie Warblers, White-eyed Vireos, Yellow-breasted Chats, American Goldfinch, Northern Cardinals, Eastern Towhees, Golden-winged and Blue-winged Warblers, and Common Yellowthroats (Brewer 1958, Chapman 1977, Crawford, et al. 1978, Whitmore 1978, LeClerc 1982, Wray 1982). Older stages of succession support bird species typically found in forested habitat, such as Red-eyed Vireo, American Redstart, Wood Thrush, Ovenbird, Carolina Wren, Downy and Hairy Woodpeckers, Kentucky Warbler, Scarlet Tanager, Carolina Chickadee, Hooded Warbler, Worm-eating Warbler, Eastern Wood-pewee, and Tufted Titmouse (Brewer 1958, Chapman 1977, Crawford et al. 1978, Allaire 1979).

Bird species also use wetlands associated with mine areas. Perkins and Lawrence (1985) found several species of waterfowl using wetlands created by surface mining in west-central Illinois, including Canada Goose, Mallard, Black Duck, Blue-winged Teal, Green-winged Teal, Wood Duck, Hooded Merganser, Lesser Scaup, Northern Pintail, Mute Swan, American Coot, Common Moorhen, and Pied-billed Grebe. Shorebird and wading species found using wetlands include Killdeer, Spotted Sandpiper, American Bittern, Green Heron, Great Blue Heron, Great Egrets, Cattle Egrets, Soras, and King Rails (Perkins and Lawrence 1985). Allaire (1979) also examined wetlands associated with mines in eastern Kentucky and observed the same waterfowl species as Perkins and Lawrence (1985), as well as Gadwalls, American Wigeons, Northern Shovelers, Redheads, Ring-necked Ducks, Common Goldeneyes, Buffleheads, and Common Mergansers. He also observed American Golden-plovers, American Woodcock, Common Snipe, Solitary Sandpipers, Greater and Lesser Yellowlegs, Pectoral Sandpipers, White-rumped Sandpipers, Baird's Sandpipers, Least Sandpipers, Semipalmated Sandpipers, and Western Sandpipers, in addition to the shorebirds and waders observed by Perkins and Lawrence (1985).

Lawrence et al. (1985) examined avian use of wetlands on reclaimed mines in Illinois and found 2 loon species (*Gavia* spp.), 2 grebe species, and many species of waterfowl, wading birds, and shorebirds on their sites. Researchers in Indiana, Illinois, Kentucky, West Virginia, and Pennsylvania also observed similar species using wetlands on reclaimed mines (Brooks et al. 1985, Krause et al. 1985, McConnell and Samuel 1985).

Bird-habitat Relationships on Reclaimed Mines

Several researchers have examined the relationship between bird abundance and habitat variables on reclaimed mines (Chapman 1977, Chapman et al. 1978, Whitmore 1979, Wray 1979, Wackenhut 1980, Strait 1981, LeClerc 1982). With the exception of Chapman (1977), all of these studies were conducted on small surface mines in northern West Virginia.

Chapman (1977) and Chapman et al. (1978) used linear regression to examine the relationship between bird abundance and 17 vegetation parameters on abandoned contour mines in southwest Virginia. They found a strong positive correlation between the percent ground cover and number of species found on mines. They also found that the number of species increased with canopy height heterogeneity, suggesting that vertical structure is an important predictor of species richness. Chapman et al. (1978) advise reclaimers to quickly establish a high degree of vegetative cover on reclaimed mines and also to provide for the development of higher vegetative strata by planting tree seedlings interspersed with herbs and shrubs.

Most of the West Virginia studies were conducted on 4 reclaimed surface mines in Preston County ranging in size from 9.1-ha to 41.5-ha. These studies examined both habitat selection and the effect of vegetative structure on reproductive success of grassland birds. Whitmore (1979) studied the effects of vegetation changes on Grasshopper Sparrows. He found that changes in bird density were due to changes in the amount of bare ground cover. As the amount of bare ground decreased below the optimum and the amount of litter cover increased above the optimum required by Grasshopper Sparrows, densities decreased. He found similar relationships for Savannah Sparrows and Vesper Sparrows, whereas Eastern Meadowlarks showed opposite trends: as bare ground decreased and litter increased their densities increased. Whitmore (1979) suggests that the density of ground cover is the key variable affecting a grassland bird's choice of a habitat patch. The birds need enough cover for nesting sites, but also need open areas for foraging, courtship, etc.

Habitat selection by Horned Larks on reclaimed mines was studied by Wackenhut (1980). Horned Larks appeared to avoid shrub cover and to prefer areas with little (12%) forb and grass cover. There were no differences in vegetative structure between successful and unsuccessful nests (Wackenhut 1980). Both Wray (1979) and Strait (1981) worked on the same mines as Wackenhut (1980) and examined habitat selection and niche separation of 3 sparrow species (Vesper, Grasshopper, and Savannah). Wray (1979) found that the vegetation around nests sites differed among the 3 species and that successful nests had more or taller vegetation than unsuccessful nests. Strait (1981) determined that Vesper Sparrow nests were associated with a greater amount of bare ground than Grasshopper and Savannah Sparrow nests. Grasshopper Sparrow nests also had a higher amount of forb cover than Savannah Sparrow nests. Vesper Sparrows preferred more open areas than the other 2 species, and vegetation surrounding Vesper Sparrow nests did not appear to affect the probability of nest predation. Successful Grasshopper Sparrow nests had less grass cover and greater forb height than unsuccessful nests. Successful Savannah Sparrow nests were associated with higher vegetative density (Strait 1981). These results indicate that sparrow species are selecting nest sites based on vegetative characteristics, that each species needs different parameters for nesting, and that nest survival depends on characteristics of the surrounding vegetation.

LeClerc (1982) examined the relationship between vegetative structure and bird species on 23 surface mines in northern West Virginia. Using discriminant function analysis she found 4 habitat variables that satisfactorily discriminated among mine sites: percent grass cover, percent bare ground, litter depth, and effective height of vegetation. She also examined bird communities by mine type and found that contour mines were distinctly different from surface mines in bird species composition. Five species were unique to contour mines: Northern Cardinals, Black-capped Chickadees, Prairie Warblers, Eastern Towhees, and White-eyed Vireos, all species typical of forest edge or early successional stages. She did not find any grassland bird species on contour mines. However, her results were confounded by time since reclamation. Her contour mines were 15+ years old, and her surface mines were <10 years old. Thus, it was not surprising that bird communities differed between these 2 mine types due to differences in vegetative structure.

LeClerc (1982) also used discriminant function analysis to examine habitat relationships among mine sites for 6 species of grassland birds. Both Savannah and Grasshopper sparrows were more likely to be present on mines with greater forb cover and minimal shrub cover and bare ground cover. Eastern Meadowlarks preferred mines with less shrub cover and vertical density and greater grass cover. Vesper Sparrows preferred mines with less grass cover, a deep litter depth, and higher forb cover and shrub cover. Horned Larks were associated with mines with low grass cover and low shrub cover, whereas Red-winged Blackbirds preferred mines with high grass cover and forb cover.

Reproductive Success of Songbirds on Reclaimed Mines

Several studies have documented the nesting success of songbirds on reclaimed surface mines in Preston County in northern West Virginia (Wray et al. 1978a, Wray 1979, Wackenhut 1980, Strait 1981, Wray 1982, Wray et al. 1982). We found no published studies of songbird reproductive success on any type of mine outside of West Virginia. A study was recently completed on large reclaimed mines in southern Indiana (Galligan and Lima, pers. comm.), but these data are currently unavailable.

All the West Virginia studies were conducted on the same mines and used the same data set. One study focused primarily on Horned Larks (Wackenhut 1980), while the others concentrated on sparrows. Wray (1978) concentrated on the reproductive biology of sparrows; Strait (1981) examined the habitat selection of sparrows, and Wray (1982) examined community structure and function on reclaimed surface mines. These researchers suggested that passerines breeding on surface mines may be double-brooded or triple-brooded, and that predation accounted for 48% of nest losses. The mean clutch size of the 4 most common nesting species in these studies (Vesper Sparrow, Grasshopper Sparrow, Savannah Sparrow, and Horned Lark) ranged from 3.20-5.25, and the probability of an egg producing a fledgling ranged from 0.05-0.32. Number of fledglings produced per hectare ranged from 0.05 to 1.45.

They found that Grasshopper, Savannah, Vesper, and Field Sparrows had clutch sizes that were similar to those published in the literature for these species, but the number of fledglings produced per hectare was lower than normally expected in natural grasslands (Wray et al. 1982). These studies examined nest losses over a 3-year period, and found that Vesper Sparrow losses remained relatively constant over the 3 years, while Grasshopper Sparrow losses increased and Savannah Sparrow losses fluctuated. They suggested that the primary predators on nests in reclaimed mine habitat were black racers (*Coluber constrictor constrictor*) and American Crows. They also found that adult sparrows did not appear to be replacing themselves sufficiently in reclaimed mine habitat and suggested that immigration is necessary to sustain a stable population. Fledging success ranged from 4.3-6.9% for Grasshopper Sparrows, from 3.6-4.8% for Vesper Sparrows, from 5.4-6.4%, for Savannah Sparrows, and was 6.6% for Field Sparrows (Strait 1981). They suggested that mines may not be a benefit to nesting sparrow species because of this poor breeding success (Wray et al. 1982).

Wackenhut (1980) examined 47 active Horned Lark nests on surface mines and found that the probability of nest survival was only 4.8%. Seventy percent of nest losses were due to depredation.

Effects of Mining on Forest-dwelling Songbirds

The major effect of MTMVF on forest-dwelling songbirds is the loss and fragmentation of forested habitat. Habitat loss and forest fragmentation have become major areas of focus in conservation biology (Harris 1984, Petit et al. 1995). It has been suggested that forest fragmentation has negative effects on the abundance, diversity, and reproductive success of forest-interior songbird populations (Finch 1991, Faaborg et el. 1995, Robinson et al. 1995). Fragmentation may negatively affect forest-dwelling songbirds because of isolation effects, area effects, edge effects, and competitive species interactions (Finch 1991, Faaborg et al. 1995).

In a forested landscape, fragmentation results from timber harvests, roads, powerlines, stand diversity, and natural canopy gaps. This is a much finer scale than occurs in agricultural areas, where forests appear as "islands" in a sea of crops and/or pastureland. Fragmentation on industrial forest might be viewed as "internal" or soft fragmentation, whereas fragmentation in an agricultural landscape might be viewed as "external" or hard fragmentation (Hunter 1990). Fragmentation in an agricultural landscape is often permanent, but fragmentation in forested landscapes is usually temporary (Faaborg et al. 1995). Faaborg et al. (1995) suggest that the latter type of fragmentation is less severe to forest birds than permanent fragmentation, but nonetheless, "detrimental effects still exist." There are no published studies documenting the effect of MTMVF on forest-dwelling songbirds as forests are lost and fragmented due to mining activities. Thus, it is unclear whether or not MTMVF acts as an internal or external fragmentation event to songbird species. However, because of the large size of most MTMVF areas, it is possible that they may have severe negative effects on populations of forest interior species that require large blocks of unfragmented forest for breeding. The severity of the habitat loss/fragmentation also will depend on whether or not MTMVF areas are re-forested or if they remain in early stages of succession. Non-timber post-mining land uses such as grazing or development will result in permanent fragmentation of forest habitats

Previous research suggests that a high amount of edge habitat might be detrimental to forest-dwelling songbird species (see Paton 1991 for a review). These studies suggest that songbirds are attracted to edges for nesting, but incur higher nest predation rates and higher parasitism rates from the Brown-headed Cowbird, a nest parasite that is known to reduce the productivity of forest songbirds. These edge effects likely only occur <25m into forest (Paton 1991). Moreover, it has been determined that higher rates of predation near edges occurred more frequently in fragmented landscapes than in forested landscapes (Hartley and Hunter 1998). Brown-headed cowbird parasitism also appears to be more detrimental to songbirds in fragmented landscapes than in contiguous forest (Donovan et al. 1995, Hagan et al. 1997). Because MTMVF creates a large amount of edge habitat, the effect on forest-dwelling songbirds must be quantified.

Raptors

We found little published literature about raptors and mining. All research found concerning the effects of mining on raptor populations involved various types of surface mining other than MTMVF. These past studies, focusing on Red-tailed Hawks, American Kestrels, and Northern Harriers, attempted to describe habitat, perch use, and nesting by raptors in and around reclaimed surface mines.

Mindell (1978) described habitat use of Red-tailed Hawks on 12 reclaimed surface mines ranging from 0.7-40 ha in northern West Virginia and southern Pennsylvania. He found that red-tails selected natural or strip-mined edge as well as intact deciduous woods, over natural or strip-mined open areas. Higher use of forest edge in proportion to its availability suggested that edge is important to Red-tailed Hawks. Mindell (1978) suggested that this was due to high prey density along both strip-mined and natural edge, greater number of perches for hunting and resting, and a greater amount of concealment cover along edges. Deciduous forest also was used more than open areas, although small mammal trapping revealed lower prey densities within the forest. He attributed the selection for deciduous forest over open areas to greater availability of resting, concealment, and nesting areas. Mindell (1978) suggested that open areas were used the least, because a majority of the area was out of visual range of the edge and had little value to Red-tailed Hawks due to lack of hunting perches. Although strip-mined habitat was used the least, immature Red-tailed Hawks were seen using these areas, possibly because of the presence of high insect populations.

Forren (1981) conducted a later study on 4 reclaimed surface mines in northern West Virginia, the largest mine being 27 ha in size. Artificial perches for raptors were constructed in reclaimed surface mines to determine if this would increase use by raptors. Use of areas with perches did increase compared to those without, but perch use was restricted to a small number of raptor species. Artificial perches were mainly used by American Kestrels (99%), and minimally by Red-tailed Hawks (0.05%) and Great Horned Owls (0.03%). Perch use peaked in the morning and evening, was highest in July and August, and 6-m perches were used more than 3-m perches. According to Forren (1981), Red-tailed Hawk and Great Horned Owl use was thought to be minor due to low detectability of small mammals in the thick vegetation found on the surface mine. American Kestrels were able to avoid this problem by preying mostly on insects, which occurred at higher densities than small mammals (Forren 1981). Insects and small mammal abundance was measured through sweep netting and trap and removal methods, respectively. Finally, examination of raptor pellets (primarily American Kestrels) showed mostly mammalian remains during May and June, but mostly insect remains during July to October, the period of highest perch use.

Yahner and Rohrbaugh (1998) compared abundance of diurnal raptors on reclaimed surface mines and agricultural habitats in both northwestern and northcentral Pennsylvania. The majority of sightings included 3 species: Red-tailed Hawks, American Kestrels, and Northern Harriers. Other species observed were Cooper's Hawk, Osprey, Broad-winged Hawk, Red-shouldered Hawk, Sharp-shinned Hawk, and Northern Goshawk. Red-tailed Hawks were commonly observed in both habitats in northwestern Pennsylvania and on agricultural habitats in north-central Pennsylvania, but less than expected on reclaimed mines in north-central Pennsylvania (Yahner and Rohrbaugh 1998). American Kestrels and Northern Harriers both occurred more than expected on reclaimed surface mines in the northwest, but American Kestrels occurred less than expected in agricultural habitats in the north-central region, whereas Northern Harriers occurred less than expected in agricultural habitats in the northwestern region. Yahner and Rohrbaugh (1998) concluded that reclaimed surface mines in the northwestern region of Pennsylvania provided suitable habitat for these 3 species, possibly by providing more breeding habitat. Another study by Rohrbaugh and Yahner (1996) used probable and confirmed breeding attempts of Northern Harriers, which were based on Pennsylvania Breeding Bird Atlas data, to correlate the number of breeding attempts in 6 regions of Pennsylvania with the number of reclaimed surface mines in the same 6 regions. They found that the number of breeding attempts by Northern Harriers in the Pittsburgh Plateau Section of the Appalachian Plateau Province were significantly greater than expected, containing 49% of all breeding attempts. This region also had a greater number of surface mines than expected, with 75% of the surface mines in the 6 regions. They concluded that Northern Harriers were associated more than expected with the open grassland habitat created after surface mine reclamation, and suggested that harriers may prefer these areas for nesting over agricultural habitats due to less disturbance associated with reclaimed mine sites (Rohrbaugh and Yahner 1996). However they did not actually locate and monitor northern harrier nests on reclaimed mines, so their conclusion is speculative.

Summary

Large-scale mountaintop removal/valley fill mining has raised questions concerning impacts on raptor populations. Several raptor species, particularly the Red-shouldered Hawk, are considered primarily forest species and breed in large tracts of contiguous, mature forest (Hall 1983, Crocoll 1994). Conversion of forest tracts to earlier successional habitats will change the raptor community in an area from predominantly forest-dependent species to open country species. Creation of fragmented forest patches may also decrease the suitability of forests remaining on or near MTMVF areas and lead to lower abundance of forest raptor populations, which tend to breed in large blocks of intact forest. Although some raptor species such as Red-tailed Hawks have shown a positive response to forest edge created by a small amount of surface mining, it is unknown whether larger areas affected by mining may dissuade use by raptors, mainly because there is proportionally less edge available, there are more open areas lacking perches, and they are more likely to be reclaimed with dense vegetation with low prey detectability (Mindell 1978, Forren 1981). Previous studies examined habitat and perch use by raptors on surface mines other than MTMVF areas (Mindell 1978, Forren 1981). We found no published studies comparing forested habitats with reclaimed areas. The fragmentation of forest and creation of edge by mountaintop removal mines may have variable effects on raptor species. Greater amounts of edge can decrease suitability of an area for Red-shouldered Hawks but increase suitability for Red-tailed Hawks (Moorman and Chapman 1996) and increase competition between these species (Bednarz and Dinsmore 1981, Moorman and Chapman 1996). Other species such as American Kestrels and Northern Harriers may benefit from open areas created by mountaintop mining, since they are often observed hunting in open areas (Bent 1937, Forren 1981), but low availability of suitable perches in open areas may limit use of reclaimed mine lands (Mindell 1978, Bloom et al. 1993). Thus, it is important to quantify what effect relatively large-scale mountaintop removal mines are having on raptor abundance, diversity, and habitat use.

Mammals

Small Mammals and Mining

Although no previous study has examined small mammal populations on MTMVF areas, there have been several studies of small mammals on strip-mined lands throughout the coal mining regions of the mid-western and eastern United States (Verts 1957, De Capita and Bookout 1975, Sly 1976, Hansen and Warnock 1978, Urbanek and Klimstra 1980, McGowan and Bookout 1986). Another study assessed small mammal populations in the Adirondack Mountains of New York on reclaimed open-pit mines for ilmenite (titanium) and magnetite (iron) ores (Kirkland 1976). The mining techniques used in these studies were considerably different from mountaintop removal mining, and the studies did not take place in West Virginia. However, they provide information on small mammal populations following a severe disturbance and subsequent reclamation.

Several studies found that small mammal communities on mines differ as a function of time after the mining activity ceased (Verts 1957, Sly 1976, Hansen and Warnock 1978, McGowan and Bookout 1986). Verts (1957) studied small mammals on 18 strip-mined sites in Illinois 4-22 years after reclamation. The mining process in the relatively flat state of Illinois is somewhat different from that used in the more topographically complex landscape of West Virginia. Verts (1957) describes the process of stripping the soil and rock overburden and then piling it behind the active mine. As the mining operation progresses, a series of parallel ridges are left behind, each about 6.1 to 9.1-m high and about 15.2-m apart. Verts (1957) focused on white-footed mice (*Peromyscus leucopus*) and prairie deer mice (*P. maniculatus bairdii*) and did not report other species captured. He found that the more recently mined areas, where the prairie deer mouse was the dominant species, had the highest overall abundance. The earliest-mined sites, where only the white-footed mouse was captured, had the next highest abundance. Lowest abundance was found on intermediate-aged sites where both species occurred in approximately equal numbers. His analysis of vegetative characteristics did not show differences in species composition, relative abundance, height of vegetation, or percentage of bare ground among the different-aged strip mines. More recently mined sites did have smaller tree diameters and tree height than the earlier mined sites. Still, the data did not support the idea that differences in *Peromyscus* species occupation of these sites was due to plant succession. Instead, Verts speculated that it was caused by differences in light, water, food, accumulated litter, temperature, and relative humidity among the various-aged strip mines.

Sly (1976) conducted a similar study in Indiana, using 3 study sites of different ages. In contrast to Verts (1957), he did not focus on any particular small mammal species, but instead tried to examine the full range of small mammal fauna. However, the only additional species he captured in significant numbers were prairie voles (*Microtus ochrogaster*). His results were similar to those of Verts (1957) in that more recently mined areas had higher overall small mammal abundances than areas that had been less recently mined. The white-footed mouse appeared to select for wooded areas, and the prairie deer mouse and prairie vole selected for areas with little or no woody cover. Hansen and Warnock (1978) and Urbanek and Klimstra (1980) also worked on Illinois strip mines. Both studies had results that were in concurrence with the studies mentioned above: small mammal abundance was higher on recently mined areas than on older areas, white-footed mouse abundance was higher in forests than mined areas, and prairie deer mouse abundance was higher in reclaimed grasslands than forests. McGowan and Bookout (1986) took a slightly different approach; they compared small mammal populations between mined areas that had been reclaimed under different regulations in Ohio. Their goal was to assess whether the passage of more stringent legislation in 1972 for the reclamation of surface mines had affected small mammals. They examined 3 previously mined areas, 2 reclaimed after and 1 reclaimed before the law change. Their results suggested that small mammals were present in greater abundance on areas that had been reclaimed after 1972 than on areas reclaimed before 1972. However, their study results were confounded by the fact that the sites on which the more stringent rules were followed had been reclaimed approximately 10 years after the site that followed the old reclamation laws, so the small mammal density difference may have been related, in part, to vegetative structure.

Each of the studies mentioned above differs from our study in a significant way. Investigators in these studies focused on comparisons among several different age classes of reclaimed mines, whereas we conducted a comparison between reclaimed areas, remnant fragmented forests, and intact forests. In other words, these studies evaluated the changes in small mammal abundance and species composition as a function of time-since-reclamation, while we compared the habitats left after mining (i.e. reclaimed grasslands/shrublands and forest patches) with relatively undisturbed areas (i.e. intact forest). Kirkland (1976) performed a study on open-pit ilmenite and magnetite ore mines in the Adirondack Mountains of New York. His approach was comparable to ours since he sampled small mammals on reclaimed mines (from 1-20 years old) and compared these results to small mammal populations in nearby intact forests. He found a significant difference in species richness between the 2 areas. Overall, 13 species were captured, but only 7 of these were found on previously mined sites, while all 13 were found in intact forests. The intact forests also had higher small mammal abundance, with the deer mouse the only species represented in significant numbers on the mined areas. De Capita and Bookout (1975) compared mined to unmined areas in Ohio. They found higher abundance of *Peromyscus* species, meadow vole, and raccoon on previously mined lands than on unmined lands. Other species, such as short-tailed shrew (*Blarina brevicauda*), opossum

(*Didelphis virginiana*), groundhog (*Marmota monax*), eastern cottontail (*Sylvilagus floridanus*) , and eastern chipmunk (*Tamias striatus*) were present in higher numbers on unmined lands. Unmined lands, in this study, included 3 different habitats: old field, old field-pine, and deciduous woods. Mined land was also of three types: brush hardwoods, hardwoods, and non-vegetated. This fact may confound the results of their study as old fields and reclaimed lands may be in similar stages of succession, having similar vegetative species composition and structure.

Urbanek and Klimstra's (1980) study also yielded results that we can compare to those of our study. Although they did not trap a control (relatively large and intact) forest as we did, they evaluated the small mammal abundance and species richness indices that they found on reclaimed mines in Illinois to those of a previous study conducted on unmined areas near their sites (Terpening et al*.* 1975). This comparison indicated that small mammal abundance was higher on the mined sites than the intact forests and that species richness was not different between the 2 areas. However, small mammal abundance can vary temporally (both yearly and seasonally), so this difference in abundance could be due to temporal rather than habitat differences.

Of the studies examining small mammals and coal mining, the most relevant to our project was a study conducted by Mindell (1978) who trapped small mammals to assess coal mines as raptor habitat in Monongalia County, West Virginia and Green County, Pennsylvania. Using snap traps on reclaimed mines ranging in size from 0.7 to 40 hectares and forests adjacent to mines, he captured 5 species, with meadow voles (*M. pennsylvanicus*) the most common, representing about 70% of the total. Other species captured were short-tailed shrew, white-footed mice, deer mice, and meadow jumping mice (*Zapus hudsonius*). He combined the 2 *Peromyscus* species for analyses because they are difficult to differentiate in this part of their range. Though these 5 species were all found on reclaimed sites, chi-square tests showed that some were more common in either reclaimed areas or forest. For example, *Peromyscus* species selected for forest whereas meadow voles selected for reclaimed areas. Mindell also found that combined small mammal abundance was higher on reclaimed mines than in forests, and that there was a significant positive correlation between litter depth and small mammal abundance among all treatments. His study, however, aimed to assess abundance of small mammals as a potential prey base for raptors, so richness was not calculated nor compared between treatments. Forren (1981) also looked at small mammals in Monongalia County, West Virginia as prey for raptors on several strip-mined areas that had been reclaimed between 1971 and 1976 and ranged in size from 16 to 27 ha; however, he did not trap in forested areas. He found the same 5 species as Mindell with meadow voles representing 56.8% of the total. Like Mindell, Forren determined that there was a significant positive correlation between litter depth and small mammal numbers.

Amrani (1987) compared small mammal populations on surface mine cattail (*Typha* spp.) marshes with populations on nearby reclaimed grasslands in West Virginia. She found that *Peromyscus* (*P. leucopus* and *P. maniculatus* combined) were more abundant in marshes than in grasslands, as was overall small mammal abundance. The marsh may provide a more favorable microclimate during weather extremes such as the heat of summer (McConnell and Samuel 1985). There was, however, no difference in abundance of meadow voles between the 2 treatments. Short-tailed shrews, meadow jumping mice, and house mice (*Mus musculus*) also were captured, but too infrequently for statistical comparisons.

Small Mammals and Forest Fragmentation

Numerous studies have examined the effects of forest fragmentation on small mammals (Gottfried 1977, Yahner 1986, Yahner 1992, Nupp and Swihart 1996, Rosenblatt et al. 1999). Gottfried (1977) compared small mammal abundance and diversity between woodlot islands and large forest tracts in eastern Iowa, and found a positive relationship between forest area and small mammal diversity and abundance. Larger forest islands may have higher diversity because there is more habitat that can support a larger population and lower the chance of a species becoming locally extinct. A second possibility is that larger forest patches are more likely to contain greater diversities of microhabitats, allowing more species to coexist (MacArthur and Wilson 1967). A positive mammalian diversity to forest area relationship also was found by Rosenblatt et al. (1999) in a study of Illinois forest patches ranging from 1.8 to 600 ha. They did not limit their study to just small mammals; instead, they looked at all mammals except bats. Sciurid species such as gray squirrels (*Sciurus carolinensis*), southern flying squirrels (*Glaucomys volans*), and eastern chipmunks (*Tamias striatus*) only were found in larger islands of forest; they did not specify, however, whether small mammal abundance differed between large and small patches. Nupp and Swihart (1996) studied white-footed mice in Indiana, comparing populations in 15 woodlots of various sizes to 3 continuous forests. They found higher densities in small woodlots as well as an inverse relationship between mass of adult male mice and forest patch size. They speculated that small woodlots may have higher food availability since trees and shrubs may be more productive at forest edges, leading to a greater supply of seeds. Also, they note that sciurid species are generally absent from small woodlots, releasing the white-footed mouse from competition for mast during autumn and winter. These results are opposite of Yahner's (1986) results in a study of the spatial distribution of white-footed mice on a forested landscape fragmented by clearcuts in Pennsylvania. Yahner suggested that white-footed mice strongly select for the interior zones of forests, possibly due to differences in predation pressures or food abundance between the forest interior and the edge zones. In a later study, Yahner (1992) examined the effects of habitat fragmentation due to forestry on small mammals in Pennsylvania, trapping on sites classified as 25-, 50-, and 75% fragmented. He found that the white-footed mouse became significantly more abundant as percent fragmentation increased.

Other Mammals

Hemler (1988) researched white-tailed deer *(Odocoileus virginianus*) use of abandoned contour surface mines in Monongalia County, West Virginia. In winter months, deer crossed mines incidentally but did not spend significant amounts of time foraging. She speculated that little use occurred because abandoned, unreclaimed mines, like a natural opening, provide little cover or food for deer. Hemler also propagated bigtooth aspen (*Populus grandidentata*) and trembling aspen (*P. tremuloides*) on these mines to evaluate this technique as a reclamation alternative. She found that deer browsed heavily on the aspen suckers in the summer months where there had been no browsing prior to the study, suggesting that aspen propagation could be a management tool to improve mines as summer deer habitat.

Knotts and Samuel (1977) also studied deer use of surface mines. They found that deer trails were common on reclaimed contour mines, following along highwalls. Heavy browsing was noted in localized areas, specifically on spoil banks that had been heavily seeded with forage species. Browsing was not found to be significant in areas 90 m or more from the highwall in the winter, which they speculated was due to the lack of cover.

Red and gray fox also used reclaimed mines. Yearsley and Samuel (1980) conducted a study in Preston County, West Virginia in which they fitted 4 gray foxes and 2 red foxes with radio collars in an area where there were patches of forest and reclaimed mines. To assess fox use of reclaimed mines in relation to other available habitats, they obtained locations on the collared animals diurnally and nocturnally. Differences in habitat use between the two fox species were not discussed. They found that fox use of mines varied seasonally, with higher use in the fall, winter, and spring than summer. The authors speculated that seasonal differences occurred because foxes feed primarily on small mammals when fruits and berries are not available, and small mammal populations were higher on the mines than in the surrounding forest. They felt that this hypothesis was supported by several observations of foxes hunting for mice on mines during these periods of high use. However, they did not sample small mammal populations.

Summary

Small mammals are an important component of biological diversity, and their populations are affected by forest fragmentation (e.g. Gottfried 1977). Further, small mammals are the primary prey base for a variety of mammalian and avian predators; thus changes in their abundance can affect other species. Although we found no previous studies of small mammal populations on MTMVF areas, there have been several studies of small mammals on strip-mined lands throughout the coal mining regions of the mid-western and eastern US (Verts 1957, De Capita and Bookout 1975, Sly 1976, Hansen and Warnock 1978, Urbanek and Klimstra 1980, McGowan and Bookout 1986). Several authors found that small mammal communities on mines differ as a function of time since mining activity ceased (Verts 1957, Sly 1976, Hansen and Warnock 1978, McGowan and Bookout 1986). Three studies compared small mammal populations on reclaimed lands with those on unmined areas (De Capita and Bookout 1975, Kirkland 1976, Urbanek and Klimstra 1980). However, results from these studies were variable with richness and abundance greater on unmined lands in 1 study (Kirkland 1976) and on reclaimed land in another (Urbanek and Klimstra 1980). Further, unmined lands in the 3rd study (De Capita and Bookhout 1975) included habitats other than intact forests which can confound results. Consequently, additional research is needed to clarify the effects of MTMVF on small mammal populations.

Herpetofauna

Amphibians are the most abundant vertebrates in many temperate forest ecosystems (Burton and Likens 1975) and make up a large part of the vertebrate biomass on certain sites (Pais et al. 1988, Heyer et al. 1994). Declines of amphibian populations have been documented throughout the world due to various causes including loss and degradation of habitats (Wyman 1990). Amphibian life-history traits make them especially sensitive to disturbances that alter microhabitat and microclimate characteristics, including physiological constraints (Feder 1983), relatively poor dispersal capabilities (Sinsch 1990), and small home ranges (Stebbins and Cohen 1995). Populations of several forest amphibian species were positively correlated with the quantity and quality of coarse woody debris, litter depth and moisture, understory vegetation density, and over-story canopy closure (deMaynadier and Hunter 1995). Gibbs (1998) suggests that amphibians may be especially prone to local extinction as a result of human-caused transformation and fragmentation of habitat due to the spatially and temporally dynamic nature of their populations. Because MTMVF alters and fragments forested landscapes, it is important to document the effects on herpetofauna, particularly amphibians.

We are aware of no published studies concerning the effect of MTMVF on the herpetofaunal community inhabiting natural hardwood/stream riparian areas. An extensive search through the West Virginia University library system, and personal communication with regional experts like Dr. T. Pauley (Marshall University) and graduate students at several Appalachian universities (California University of Pennsylvania, Marshall University, and West Virginia University) turned up little published work involving reptiles and amphibians and any form of mining. Four published studies examined the herpetofauna inhabiting ponds on surface mines (Riley 1952, Myers and Klimstra 1963, Turner and Fowler 1981, Fowler et al. 1985), and a graduate student at Marshall University (Huntington, West Virginia) is currently in the process of completing an MS research project concerning MTMVF and herpetofauna (Dr. T. Pauley, pers. comm.).

Riley (1952), examined the effect of surface mining on the regional ecology of the Midwest. His work involved very little, if any, experimentation and mainly used observational data to generalize how mining impacts vegetation and wildlife. He did, however, make reference to a few reptile and amphibian species found in midwestern surface mine ponds. Five amphibian species (America toad *Bufo americanus,* green frog *Rana clamitans,* leopard frog *R. pipiens,* pickerel frog *R. palustris*, and cricket frog *Acris crepitans*), and 3 reptile species (snapping turtle *Chelydra serpentina,* painted turtle *Chrysemys picta*, and northern water snake *Natrix sipedon*) were collected in Ohio strip mine ponds. Additionally, bullfrogs (*R. catesbeiana*) were being raised commercially in at least 1 Illinois strip mine pond. No mention is made of how these findings compare to the herpetofaunal community in undisturbed areas in that region.

Meyers and Klimstra (1963) conducted their work in Perry County, Illinois on sites that had been contour mined. The mining activities in this area left alternating ridges and valleys (spoil banks) with fairly steep slopes (45%). This topography encouraged the formation of many temporary and permanent ponds that had been colonized by a variety of plant and animal life since mining activities ceased approximately 20 years before the study was conducted. A general search (hand capturing and visual observation) found 32 species of herpetofauna inhabiting the site, but only 10 were commonly encountered. The searches were not time- or area-constrained, thus no relative abundance or population estimates were calculated. Myers and Klimstra (1963) compared the 32 species they found with the 39 (Meyers 1957) and 54 (Rossman 1960) species reported by 2 separate inventories of unmined sites located within 75 miles of their Perry County, Illinois site. They concluded that strip-mined lands in general would be inhabited by plants and animals adapted to environmental conditions produced by mining, and that additional population and/or successional studies would provide useful information.

Turner and Fowler (1981) conducted a fairly thorough search of 24 ponds on a surface mine in Campbell County, Tennessee. Because mining had ceased in 1972, the ponds were at least 6 years old when sampling was conducted in the spring of 1978. Dip nets were used to sample amphibian eggs, larvae, and adults. A students t-test was used to compare the average number of species found in ponds with different pH values. Water quality and aquatic vegetation also were sampled. Twelve of the 17 species expected to be found in the area were captured. Significantly more species ($P < 0.05$) were found in ponds with higher pH. In addition to pH, Turner and Fowler (1981) mention that water hardness and presence of emergent vegetation seemed to influence whether or not some species inhabited a particular pond. The spring peeper (*Pseudacris crucifer*) was the most commonly captured amphibian and inhabited 16 of the 24 ponds. They believe that their findings provide justification for leaving mine ponds in place after cessation of active mining, because permanent water usually provides wildlife habitat and it costs less to leave a pond than to remove it.

Fowler et al. (1985) sampled the herpetofaunal community on 11 newly constructed surface mine sediment ponds on 2 separate mines in Campbell county, Tennessee. In addition to reptiles and amphibians, water quality, invertebrates, vegetation, and fish also were sampled. Amphibians were sampled with auditory surveys on 11 surface mine ponds from 1 March 1979 to 29 February 1980. Observers also identified egg masses and used a hand-held D-net to capture larval amphibians. Twelve of the 17 species of amphibians, known to breed locally in ponds, were detected. All ponds had at least 1 species. They also found that the water quality, in most cases, was of sufficient quality to support aquatic life. Apparently, searches were not time- or area-constrained so density and/or abundance were not calculated. Fowler et al. (1985) recommended the retention of these sediment ponds after mining stopped because they seemed to have a large potential for fish and wildlife.

None of these studies were conducted on MTMVF areas, they generally did not include terrestrial species, nor did they use methods that accurately quantified time and effort. Although 3 of the studies compared the number of species found to the number of species thought to inhabit the region, no direct comparisons were provided because intact habitats were not sampled. Based upon these limited data, it seems that some herpetofauna, particularly those associated with bodies of standing water, colonize surface mine sites when mining ceases or suitable habitat is provided, however it is not known if abundance or species composition is similar to unmined habitats. These studies may indicate a general trend, but their results cannot be extrapolated to how MTMVF may affect West Virginia reptiles and amphibians due to limitations studies imposed by the methods used, lack of experimentation, and geographic and temporal differences.

Summary

Herpetofauna, particularly amphibians, can be ideal indicators of how well reclamation efforts have succeeded because they are susceptible to small environmental changes (Jones 1986) and make up a large part of the vertebrate biomass on certain sites (Pais et al. 1988, Heyer et al. 1994). However, a thorough literature search revealed little previous research concerning the effects of surface mining on herpetofauna. Myers and Klimstra (1963) and Fowler et al. (1985) studied the colonization of surface mine sediment ponds by herpetofauna, but we found no published literature regarding the effect of surface mining on stream, riparian, or terrestrial herpetofauna. Because the conditions resulting from mountaintop mining and subsequent reclamation are dramatically different from those provided by the original intact forest, more information is needed on how hepetofaunal populations are responding to these changes.

Methods

Study Areas

Study sites for the terrestrial study were selected to overlap as much as possible with study sites used for the aquatic studies. The Environmental Protection Agency (EPA) aquatic team initiated aquatic studies on 5 watersheds (Mud River, Spruce Fork, Island Creek, Clear Fork, and Twentymile Creek). Two of these watersheds (Island Creek and Clear Fork) were inappropriate for use in the terrestrial wildlife studies. Human activities on Island Creek such as grazing, orchards, and homes would have confounded study results. Clear Fork was not suitable because much of the area was reclaimed recently and little vegetation had become established. Therefore the remaining 3 watersheds were used for the terrestrial study areas (Fig. 1) in summer 2000. Initial work on the study in 1999 focused primarily on the Mud River and secondarily on the Spruce Fork watersheds.

Study areas included 4 treatments: intact forest, fragmented forest, young reclaimed mine (grassland), and older reclaimed mine (shrub/pole) (Table 1). The latter 3 treatments resulted from mining and reclamation activities. Intact forest sites are relatively large intact forested areas undisturbed by mining activities and located near the reclaimed sites, either within the same watershed as a mining site or in an adjacent watershed. Although these sites are relatively contiguous forest, they do have some breaks in canopy cover from streams, roads, and natural canopy gaps. Some intact forest sites are located in close proximity to MTMVF areas, but no intact forest site shares more than 1 edge with an MTMVF area. On the other hand, we defined *fragmented* forest as a tract of forest primarily surrounded by reclaimed mine land on at least 3 sides. Young reclaimed mine areas (grassland) consist mostly of grasses and are about 5-19 years of age. Older reclaimed mine areas (shrub/pole) contain shrub and pole-sized vegetation and are about 13-27 years of age. Because these 2 treatments are defined by vegetation characteristics of early and later successional stages, lack of succession on some older grassland sites resulted in an overlap in age for these 2 treatments. Mine ages were determined from the estimated year sites were reclaimed and were provided by Arch Coal and Cannelton Mining companies.

The intact and fragmented forest areas were comprised mostly of mature hardwood species including red oak, white oak, black oak, pignut hickory, bitternut hickory, shagbark hickory, tuliptree, American beech, red maple, sugar maple, American sycamore, white ash, and black birch (scientific names of tree and shrub species are found in Appendix 2). Understory trees (seedlings, saplings, and poles) in these areas included American beech, black birch, black gum, flowering dogwood, ironwood, red and sugar maple, sourwood, spicebush, and white ash as well as other common hardwood species. These stands were second growth forests that appeared to be approximately 60-80 years old. Although forested, these stands may have been periodically disturbed over the last several decades from firewood cutting, single tree harvesting, thinning, and forest fires.

The primary vegetation on the young reclaimed mine areas included tall fescue (*Festuca arundinacea*), sericea (*Lespedesa cuneata*), autumn olive, black locust, European black alder, and scotch pine. Vegetation on older reclaimed mine areas included goldenrod (*Solidago* spp.), tall fescue, sericea, autumn olive, black locust, scotch pine, red maple, American sycamore, tuliptree, multiifora rose, and blackberry/raspberry. Tree and shrub species on these older sites were larger and more predominant than on younger sites.

Study areas included 3 MTMVF sites and nearby forest lands in southwestern West Virginia (Table 1 and 2, Fig. 1). Sample points were placed along and surrounding 15 stream drainages on the 3 watersheds (Table 1, Fig. 2-11). All figures also show locations of EPA water quality sampling points.

The Hobet 21 mine is located in the Mud River and Little Coal River watersheds in Boone County (Fig. 1 and 2). Fragmented forests on this site are forested areas surrounded on 3 sides by grassland habitat (Fig. 3). First-order streams had valley fills, whereas second-order streams were left intact. The intact forest treatment sites were located in 3 drainages; 2 were just south of the mine (Fig. 2 and 3) and 1 was located approximately 5 km northeast of the mine along the Big Buck Fork of Hewitt Creek (Fig. 4 and 5). Two areas were used for the shrub/pole treatment: 1 in the northeastern section of the mine (Fig. 2 and 3), and 1 along a valley fill at the head of the Hill Fork of Hewitt Creek (Fig. 4 and 5). All grassland sampling points were located on the mine.

The Daltex mine is located in the Spruce Fork watershed in Logan County (Fig. 1 and 6). Fragmented sites were located along a second order stream that is surrounded by reclaimed mountaintop mines and contour mines (Fig. 7). The intact forest treatment sites were located approximately 1.6 km northeast of the mine along Bend Branch of Spruce Fork, and approximately 1.6 km east of the mine along Pigeonroost Branch (Fig. 6 and 7). No shrub/pole treatment was established at Daltex because the small amount of this habitat that was available was not created by MTMVF but contour mining. All grassland sampling points were located on the mine.

The Cannelton mine is located in the Twentymile Creek watershed along the border of Kanawha and Fayette Counties (Fig. 1 and 8). The forest fragment treatment on this site was a forested areas surrounded on 3 sides by grassland habitat (Fig. 9). Intact forest sampling points were located northeast of the mine along the Ash Fork of Twentymile Creek on the border of Clay and Nicholas counties (Fig. 10-11). The EPA had selected Neil Branch, located just east of Ash Fork, as their intact site; however, recent logging activity precluded our use of this drainage. Both the grassland and shrub/pole treatments were located on the mine.

Selection of Sampling Points

Sampling points were established within each treatment at least 75 m from the edge of any other treatment and at least 250 m apart. Within the 2 forest treatments, sampling points were located 35 m from streams (to coincide with mammal transects and herpetofaunal arrays), upslope at least 75 m from streams (Fig. 12), and on or near a ridge top. Within reclaimed areas, points were positioned similarly but relative to the rip-rap channel. Sampling points were distributed over the 3 watersheds and 4 treatments (Table 2). Elevations of sampling points ranged from 241-566 m (Table 3).

Intact Forest

Points in intact forest sites were established along first- and second-order streams, with points placed 35-m from streams, 75-m upslope from streams and on or near the ridge top at the head of hollows. Sampling points were located systematically with the first point placed 75 m from an edge and 35 m from streams. Subsequent points were placed 250 m apart, alternating banks if possible. In some cases, consecutive points were on the same bank if minor edges from canopy openings or trails were present on the opposite bank. An attempt also was made to alternate consecutive points so that 1 was 35 m from the stream and the next was upslope at least 75 m. Again, this was not always possible due to the presence of edges or human disturbance. Generally we attempted to place points in the least disturbed areas, to minimize effects of edges, and to sample sites with a gradient of elevations that could be compared to head-of-hollow fills on reclaimed sites and fragmented forests along lower reaches of streams.

Fragmented Forest

The majority of fragmented forest sites occurred at the base of head-of-hollow fills (e.g. Fig. 3); therefore, the first sample point was placed 75 m from the forest/reclaimed edge and 35 m from the stream with successive points placed as described for intact forest. Fragmented forest was limited in the Spruce Fork watershed, thus points were established in what was available. Three points were placed on the south bank of Beech Creek, 2 at 35 m from the stream and 1 upslope (Figs. 6 and 7). This fragment is very narrow and the north bank was close to the road edge. The other 3 points were placed in fragments of upland forest at least 75 m from roads and other edges. At the Twentymile Creek fragment site (Hughes and Jim Forks), 6 points were established as described above along the main creek and 4 points were established along streams below head-of-hollow fills that drain into Hughes Fork (Fig. 9). Fragments with sampling points ranged in size from 30-214 ha (Table 3).

Reclaimed Grasslands

At the Mud River and Twentymile Creek sites, we placed 1 point 35 m from rip-rap channels in head-of-hollow fills on reclaimed grassland sites, and remaining points were placed upslope in areas above valley fills to sample areas of higher elevation. These latter points were not positioned relative to the channel, but were kept 250 m apart. At the Spruce Fork site (Rockhouse Creek), 6 sampling points were established along the main rip-rap channel of Rockhouse Creek, alternating banks and distances from channels. Another 6 plots were located above the valley fill on the top of the mountain. The estimated age of grassland points ranged from 5-19 years (Table 3).

Reclaimed Shrub/pole

Shrub/pole points were established at Twentymile Creek and Mud River sites. This treatment was limited, and thus our points were established without regard to streams or elevation. They were placed wherever this habitat occurred, and where points could be placed at least 75 m from the edge and at least 250 m apart. Six sample points, at the Cannelton mine, were placed in an area that we were told was the oldest MTMVF site in West Virginia. The age of shrub/pole points ranged from 13-27 years (Table 3).

Songbird Abundance

Songbird abundance was measured from 0630 to 1030 hrs on fixed-radius 50-m point count plots using standardized methods (Ralph et al. 1993). All birds seen or heard in a 10-min period were recorded. We recorded if the bird was observed visually or aurally, identified the sex if possible, whether it was flying over, and whether it was within or outside the 50 m plot. Surveys were not conducted during windy or rainy weather. Percent cloud cover and wind speed were recorded using standardized scales (Martin et al. 1997, Table 4). All point counts were surveyed twice during the breeding season (late May-June), each time by a different observer. Points were surveyed twice in order to increase the number of species detected. Petit et al. (1995) determined that 20% more bird species are detected with 2 counts than with 1 in eastern deciduous forests, and that 20 min of total counting time (two 10-min counts) is required to develop a relatively complete species list. Two observers conducted all counts in 1999; these 2 individuals plus a third person conducted all counts in 2000. All observers had previous experience identifying songbird species by sight and sound. Prior to initiating surveys, observers conducted simultaneous point counts to verify bird identification skills and distance estimation. At least 3 practice sessions in each habitat type (grass, shrub/pole, and forest) were conducted. After conducting the point counts, observers compared species and distances estimated. Observers then paced 50 m in order to improve their distance estimation skills. They also paced to approximate locations of different bird species to practice placement of birds within or outside the 50-m radius circle. The maximum number of birds at each count was used in data summaries and analyses. Each sampling point station was geographically referenced using a global positioning system (GPS).

Songbirds were placed into 1 of 4 habitat guilds based on their habitat preferences and into 1 of 5 nesting guilds based on where they place their nests. Habitat guilds were: grassland, edge, interior-edge, and forest interior. Nesting guilds were: ground, shrub, subcanopy, canopy, and cavity. Birds were placed into these guilds and groups based on Whitcomb et al. (1981), Ehrlich et al. (1988) and from personal observation of species in the study area. Abundances of each guild were compared among treatments using a two-way analysis of variance (ANOVA) with treatment and year as factors (Zar 1999). If a treatment by year interaction occurred, we conducted one-way ANOVA tests comparing treatments in each year separately. Total abundance and species richness also were compared using ANOVA.   The Waller-Duncan k-ratio t-test was used to examine differences between individual treatment means. Additionally, individual species that were observed at >5% of point counts in fragmented and intact forest were tested for differences using ANOVA between fragmented and intact forest. We also used the Jaccard and Renkonen indices to examine community similarity between pairs of treatments (Nur et al. 1999). Bird species that are typically difficult to survey with point counts, such as flocking species, species with large territories, and non-vocal species, were excluded from the analyses of total abundance, species richness, and similarity. Bird abundances and guild abundances were transformed prior to analyses using the transformation $X'=\log_{10}(X+1)$, where X' is the transformed value and X is the original value (Zar 1999). Although most abundances were not normally distributed after transformation, we chose to proceed with ANOVA because ANOVA is "robust with respect to the assumption of the underlying populations' normality" (Zar 1999). Avian nomenclature follows the American Ornithologists' Union Check-list of North American Birds, seventh addition (AOU 1998, Appendix 1).

Partners in Flight (PIF) identified 15 songbird species as priority species for conservation in the upland forest community of the Ohio Hills and Northern Cumberland Plateau physiographic areas, the 2 areas within which our study sites fall (Table 5; Rosenberg 2000, R. McClain, personal communication). The Cerulean Warbler in particular is listed as being at Action level II (in need of immediate management or policy rangewide) by PIF. The Louisiana Waterthrush and Eastern Wood-pewee are other species of concern, listed at Action level III (management needed to reverse or stabilize populations). The other 12 species are at Action level IV (long-term planning to ensure stable populations needed). We developed logistic regression models for the 11 listed species (Cerulean Warbler, Louisiana Waterthrush, Worm-eating Warbler, Kentucky Warbler,Acadian Flycatcher, Wood Thrush, Yellow-throated Vireo, Hooded Warbler, Scarlet Tanager, Black-and-white Warbler, and Yellow-billed Cuckoo) that were found at >5% of point counts (Table 5).

We used forward logistic regression (Neter et al. 1996) to examine the relationship between habitat characteristics and the presence/absence of these 10 forest songbirds using habitat data from fragmented and intact forest point counts. The significance level chosen for entry and retention in the model was 0.10. We used presence/absence as the dependent variable because at most point counts only 1 individual of a species was detected within 50 m (Hagan et al. 1997). This technique was chosen because it has been used by other researchers examining the effects of landscapes on songbird species (Hagan et al. 1997, Villard et al. 1999), and because predictor variables do not need to follow a joint multivariate normal distribution (Neter et al. 1996). The Hosmer and Lemeshow goodness-of-fit test was used to determine if the data fit the specified model. Models were rejected if the p-value for the goodness-of-fit test was <0.10, indicating that we should not reject the null hypothesis that our data fit the specified model (Cody and Smith 1997).

Nest Searching

Nest searching was conducted in 2 grassland areas on each of the 3 mines for a total of 6 sites. To obtain a good estimate of species-specific nest survival, a minimum of 20 nests per species must be monitored (Martin et al. 1997). Therefore, we set a target of 20 nests for each of the most common species in the grassland habitat (i.e. Grasshopper Sparrow and Eastern Meadowlark). However, breeding birds in grassland habitat often have low densities, and we were not able to locate this many nests by searching a defined area (plot). Thus, a plotless nest searching method was used (Martin et al. 1997) so that a larger area could be searched for breeding birds. The amount of area actually searched for nests was estimated using GIS maps of each mine site.

Each nest searching area was searched every 3 days by 2-3 field technicians trained in proper searching and monitoring techniques (Martin and Geupel 1993). Nest searching began one-half hour after sunrise and concluded 8 hr later (approximately 0600-1400 EST). Nest searching methods followed national BBIRD (Breeding Biology Research and Monitoring Database) protocols (Martin et al. 1997). Nests were located by flushing females, by following adult birds, and by observing parental behavior (i.e. carrying nest material or food, copulation). When time allowed, other project personnel also searched for songbird nests.

All nests found were monitored every 3-4 days (Martin et al.1997). Because nests in grasslands are typically well-concealed, they were marked for relocation using 2 flag stakes. The stakes were placed on either side of the nest at a distance of 15 m. Care was taken when monitoring the nest to avoid disturbing the female. When possible, nest searchers observed the nest from a distance of no less than 15 m for up to 30 min to confirm that it was still active. The nest was approached and checked for contents a maximum of 4 times: once when it was initially found, once to confirm clutch size, once to confirm brood size, and once to confirm fledging success or failure. Nests were not approached when avian predators (e.g., American Crows and/or Blue Jays) were observed nearby because these birds will follow humans to nests (Martin et al. 1997). Observers also continued to walk in a straight line after checking nest contents to avoid leaving a dead-end scent trail directly to the nest that might be followed by mammalian predators (Martin et al. 1997). The vegetation concealing nests was moved to the side using a wooden stick to avoid putting human scent on nests if the vegetation blocked the observer's view of contents.

A nest was considered successful if it fledged at least 1 young. Fledging success was confirmed by searching the area around the nest for fledglings or for parent-fledgling interactions. However, if no fledglings were observed, the nest was considered to have fledged young if the median date between the last nest check when the nest was active and the final nest check when the nest was empty was within 2 days of the predicted fledging date (Martin et al. 1997). Nest survival was calculated using the Mayfield method (Mayfield 1961, Mayfield 1975). Daily nest survival estimates were calculated for the incubation and brooding periods separately because nest survival may differ between these 2 periods. The overall daily survival rate was calculated as the product of incubation and brood daily survival. Survival during the egg-laying stage was not included in the calculation of overall nest survival because we found few nests during this stage of the nesting cycle.

Surveys to determine fledgling density were conducted in late July and early August on each mine. Three 500-m transects on each mine were walked at a pace of 1.5 km/hr and all fledglings seen within 25 m of either side of the transect center line were recorded. Transects were established to coincide with areas that had been searched for nests. Fledgling densities were determined by calculating the number of fledglings divided by 2.5 ha (i.e 500 m x 2(25) m) on each transect. The average of the 3 transects was used as the measure of fledgling density for each mine.

Bird and Mammal Use of Ponds

In summer 2000, we documented presence/absence of small mammals and birds that used ponds located on reclaimed mine sites during early May, late June, and late August (mammals), and early May, late June, and late September (birds). Sample dates for mammals were selected to coincide with the new moon because small mammals are more active when the moon is dark. Ponds on each mine were identified using aerial photographs and ground truthed for accuracy. Ponds were placed subjectively into 2 size classes, either small or large. Ten ponds in each size class, for a total of 20 ponds, were selected randomly and distributed over the 3 mines. Small ponds averaged 0.16 ha (range:0.03-0.28 ha), and large ponds averaged 0.53 ha (range: 0.30-1.38 ha). We placed a small mammal trapping transect 100 m in length within 10 m of each pond margin. Two Sherman live-traps placed at each of 10 trapping stations spaced 10 m apart along the transect were baited with a mixture of peanut butter and rolled oats. Traps were open for 2 nights during each sample period. All animals captured were marked and released. All birds observed using the pond were recorded as field technicians were approaching the pond and during a 10-min point count. At each pond, we established a bird point count station on the side of the pond opposite the small mammal transect. All birds seen or heard within 50 m of the pond were recorded using standard point count methods described above. Mammal and bird data from pond surveys were used only to document presence/absence.

Vegetation Measurement

All Treatments

We measured vegetation and habitat characteristics on all sampling points within each treatment using methods modified from James and Shugart (1970) and the Breeding Bird Research Database program (BBIRD; Martin et al. 1997). Within each point count circle, 4 0.04 ha vegetation subplots were established (Fig. 12). Subplots were placed at the center of the circle, and 35 m away at 0°, 120°, and 240°. At points associated with small mammal transects, 2 subplots were located on the transect line, 1 centered on the point count, and 1 upslope from the point count center. Subplots along the mammal transect were located 45 m from the center and spaced approximately 60 m from each other (Fig. 13). The upslope plot remained 35 m from the center.

Within each 0.04 ha subplot, all tree species were identified and placed into 1 of 5 diameter-at-breast height (dbh) classes: >8-23 cm, >23-38 cm, >38-53 cm, >53-68 cm, and >68 cm. Within a 5.0-m radius circle centered on the subplot, we counted number of sapling stems (woody species >0.5 m high) in 2 size classes: ≤2.5 cm at 10 cm above ground and >2.5-8 cm at 10 cm above the ground.

An ocular sighting tube was used to measure percent ground cover and canopy cover (James and Shugart 1970). The sighting tube was a 5.0-cm pvc pipe with cross-hairs at 1 end. If the cross hairs sighted on vegetation, then canopy cover was recorded as present (a 'hit'). Five sight-tube readings were taken on each subplot every 2.26 m along 4, 11.3-m transects that intersected at the center of the subplot (Fig. 12). The number of hits divided by 20 provided a quantitative measure of percent cover. Ground cover was recorded as the cover type in the cross hairs, either green (grass, shrubs, fern, herbaceous vegetation combined), bareground/rock, moss, woody debris, water, or leaf litter. On grassland vegetation points, green vegetation was separated into more detailed categories including: grass/sedges, forbs (herbaceous plants), and shrubs (woody species <0.5 m tall). We defined woody debris as any dead woody material $\geq$4 cm in diameter on the ground. All other woody material on the ground counted as litter. Water was recorded as ground cover if the sampling point fell across a stream or pool. Canopy cover was recorded for 6 layer classes representing shrub, sapling, understory, subcanopy, codominant, and dominant trees: 0.5-3 m, >3-6 m, >6-12 m, >12-18 m, >18-24 m, and >24 m. A structural diversity index, which takes into account the amount of canopy cover in each layer class and the number of layers present, was calculated using these variables (Nichols 1996). Canopy cover and structural diversity was only measured in the shrub/pole, fragment, and intact forest treatments.

Average canopy height and percent slope were measured with a clinometer, whereas a compass was used to determine the aspect. Elevation was determined using digital elevation models in a GIS.

Edge types represented abrupt changes in habitat and may or may not have been linear (roads, streams, etc.). We identified several potential edge types on the study areas, some of which we considered "internal" edges and some that were "external" edges. Internal edges represented relatively minor breaks in continous habitat and were usually linear. External edges were usually much larger in extent than internal edges and represented a considerable break in the habitat. In intact and fragmented forest, internal edges included streams, roads, and natural gaps, and external edges included valley fills and grasslands in mined areas. In grassland and shrub/pole habitat, internal edges included roads, valley fills, ponds, and blocks of autumn olive, and external edges were primarily forest.

We recorded 3 edge classes and determined the distance of each edge from the point count center. First, the closest internal or "minor" edge type (Table 4) and distance was recorded for each subplot. The distance to this edge was determined by pacing. The average distance of the 4 subplots from any minor edge was used in analyses as the distance from minor edge. We also calculated the percentage of subplots in each treatment that were closest to the 13 minor edge types. Second, we determined the distance from the center of each point count to the closest "habitat" edge using aerial photographs in Arcview GIS. The edge types for this edge class were: grassland-shrub/pole; forest-grassland; forest-shrub/pole, and forest-active mine. Third, we calculated the distance to the closest "mine" edge (either grassland, shrub/pole, or active mine) for forest points and the distance to the closest forest for grassland and shrub/pole points. In most cases the habitat edge and the mine/forest edge were identical, but in some cases an alternative habitat was closer than the mine/forest edge.

Slope aspects were transformed before analyses the Beers et al. (1966) procedure, using the equation:

$$A' = (COS(45-A)+1)$$

where A' is the transformation index and A is the direction the slope faces in degrees (Frazer 1992). With this transformation, northeastern facing slopes receive a value OF 2 and reflect mesic conditions, while southwestern exposures receive a value of 0 and reflect xeric conditions. Other exposures are distributed between these values. We assigned an aspect index of 0 to points on dry ridge tops, and an index of 2 to points in flat bottomlands because ridge tops and bottom lands have no slope and thus no aspect, but ridge tops tend to be xeric while bottomlands are mesic (Frazer 1992).

All percentage variables (i.e. slope, ground cover, and canopy cover) were transformed using the arcsine-square root transformation (Zar 1999) prior to analyses. Stem densities were transformed using the transformation $X' = \log_{10}(X+1)$, where X' is the transformed value and X is the original value (Zar 1999).

Habitat variables were tested for differences among treatments using two-way ANOVA (Zar 1999). Treatment and mine were the main factors in the models, and treatment by mine was included as an interaction term. The average values for all variables from the 4 subplots were used in analyses. ANOVA was used to compare treatments after variables had been transformed. Similar to analyses of songbird abundances, most habitat variables were not normally distributed after transformation, but we chose to proceed with ANOVA because it is robust to deviations from normality (Zar 1999). If there was a significant interaction ($P<0.05$) between mine and treatment, we conducted one-way ANOVA's to determine the exact nature of the interaction.

Grassland and Shrub/pole Treatments

Additional vegetative measurements were collected at grassland points. A Robel pole, described below, was used to record most of these data and was used to determine the amount of vegetative cover and grass height.

The Robel pole (Robel et al. 1970) was a stick demarcated at half-decimeter intervals (Fig. 14). The pole was placed vertically on a point. An observer moved 4 m away from the pole, and with their eyes 1 m above the level of the ground, noted the lowest interval on the pole that was not completely obscured by vegetation. This interval was recorded as the distance in decimeters from the ground to the bottom of the interval. Measurements with Robel poles have been widely used to characterize vegetation around nests of birds (Kirsch et al 1978). They are used to measure height of vegetation and provide an index of biomass (Robel et al. 1970). To quantify vegetative cover, measurements with the Robel pole were taken at the subplot center, and at 1, 3, and 5 m along each transect (Fig. 15) for a total of 16 measurements. We took 4 measurements at the center, with the observer facing towards the center of the subplot from each of the 4 transect directions. A single measurement was taken at every location away from the center with the observer facing towards the center of the subplot. Vegetative cover at a point was the average of these 16 measurements.

Maximum height of herbaceous vegetation was measured to the nearest 0.5 dm (Fig. 14) using the Robel pole placed at the following locations: the center, 1, 3, 5, and 10 m along each transect (Fig. 15). At each of these locations, the height of the tallest herbaceous vegetation within a 3.0-dm radius circle of the pole was recorded. Vegetation height for the plot was the average of the 17 measurements.

The depth (in centimeters) of organic litter was measured at 13 locations along the 4 transects:

at the center and at distances of 1 m, 3 m, and 5 m along each transect (Fig. 15). If the point landed on a rock or log, we moved our measurement location to the nearest point that had mineral soil on which litter could potentially rest. If a point fell on bare ground, litter depth was recorded as 0.0 cm. We measured litter depth using the metric ruler on a compass.

Vegetation variables measured at grassland points also were measured at Grasshopper Sparrow nests in 2000. However, results were not analyzed statistically because of small sample sizes.

Raptor Abundance

Raptor abundance and habitat use were quantified at 48 of the songbird point count stations on the study areas. Stations were located approximately 0.8 km apart according to the protocol suggested by Fuller and Mosher (1987). Twelve survey stations were sampled monthly (February - September 2000) in each of the 4 treatments with roughly equal numbers of sample points over the 3 mines (Table 2). All 48 points were sampled over a 4-6-day period. Points from at least 3 treatments were sampled on a given day to minimize temporal variability between treatments. The order that points were sampled on a given day was randomly established during the first survey. On subsequent surveys, the order in which points were sampled was systematically varied through 3 daily time periods: early, mid-, and late-day.

We used broadcast surveys to sample raptor populations because broadcasting conspecific vocalizations is an effective way to survey targeted raptor species (Rosenfield et al. 1988, Mosher et al. 1990, Kennedy and Stahlecker 1993). During winter months, broadcast surveys were conducted from one-half hour after sunrise until 1600 hrs because raptors can be active throughout the day during cooler weather. During summer months, broadcast surveys were conducted from one-half hour after sunrise until 1300 hrs, because shifts in raptor activity in the afternoon may reduce the detectability of certain raptor species such as Red-tailed Hawks and *Accipiters* (Bunn et al. 1995).

Broadcast surveys lasted 10 min, and consisted of 5 min of broadcasting vocalizations and 5 min of observation/listening time. Six calls were broadcast for a 20-sec duration at 1-min intervals (20 sec of vocalization, followed by a 40-sec listening period), leaving a final listening period of 4 min and 40 sec and thus making a total of 10 min. The broadcast speaker was held 1.5 m above the ground and rotated 120° between each broadcast. Calls were broadcast at a volume of about 110 db at 1 m from the megaphone speaker. Both Great Horned Owl and Red-shouldered Hawk vocalizations were used during the survey period. The 6 vocalizations alternated between Great Horned Owl and Red-shouldered Hawk calls. Previous studies (Mosher and Fuller 1996, McLeod and Anderson 1998) have shown that many raptor species respond to either Great Horned Owl or conspecific calls. Red-shouldered Hawk vocalizations were used to specifically elicit responses from Red-shouldered Hawks (a migratory nongame bird of management concern in the Northeast; Peterson and Crocoll 1992), while the Great Horned Owl vocalizations were used to elicit responses from other raptor species. We randomly determined which type of call (Great Horned Owl or Red-shouldered Hawk) would start the first survey each month, with the second survey starting with the call not previously used, and thus alternating throughout the entire survey session each month.

Two observers trained in identification of raptors by sight and sound were present at every survey. One individual was the primary observer and was present at each survey. The second observer alternated between a number of individuals. During the 10-min survey period, both observers actively watched and listened for raptors. Surveys were not conducted in inclement weather (moderate to heavy rain, fog, or wind).

Data recorded on surveys included weather conditions (cloud cover/precipitation, wind, and temperature), nearest edge type, distance to edge, latency (time from start of survey until first raptor detection), general vegetative cover characteristics (size class of trees, amount of cover, dominant plant species), raptor species detected, age and sex (if possible), behavior during detection (perch and call, flyby and call, silent perch, silent flyby, vocal only), time each individual bird is seen, estimated distance bird is from observer, and habitat type in which a bird is first detected. Survey data were summarized as mean number individuals detected within a season. The winter season was defined as December-March, the summer season April-July, and the migration season August-November.

Roadside surveys also were conducted once in late July on each of the 3 mines. These surveys consisted of driving a specified route at 16 km/h through grassland, shrub/pole, and fragmented forest treatments, while looking and listening for raptors. The intact forest treatment was not included in roadside surveys because this treatment had no drivable roads. Each roadside survey period was similar in time and length (about 2 hrs for 16-24 km) and covered approximately equal areas of the 3 habitat treatments for each mine. The only exception was the Daltex mine, which lacked areas representative of the shrub/pole treatment. All raptor species observed were recorded along with the time, distance away from the road (m), habitat, and behavior. Other data recorded were the length of survey (km), start and end of survey, and weather conditions (cloud cover, precipitation and wind).

Small Mammal Abundance

In May-August 2000, small mammal abundance and richness were quantified on 38 150-m long transects adjacent to riparian zones with each of the 4 treatments replicated 8-10 times (Tables 1 and 2). In May-August 1999, 24 transects in 3 treatments (grassland, fragmented forest, intact forest) were sampled. The number of transects sampled for the Mud River watershed was greater than that for the other 2 watersheds because these transects had already been established and sampled in 1999 before the study was expanded to include the Twentymile Creek watershed. Small mammal transects coincided with a randomly selected subset of the songbird point count stations located 35 m from the stream or rip-rap channel. Transects crossed the 50-m radius circle of the point count plot, about 10 m from the channel (Fig. 13) and were oriented so that their centers aligned with the center of the point count station. Transects followed a constant bearing for as long as the channel allowed, changing direction only when necessary to maintain a fairly uniform channel distance. Trapping stations were placed at 10-m intervals along each transect line, with 2 Sherman live traps (7.7 x 7.7 x 23 cm) placed within 2 m of each trapping station. Thus, each transect had 30 traps. Bait consisted of a peanut butter and oat mixture. Trapping methods followed those of Jones et al. (1996).

The 38 transect lines were divided into 5 trapping blocks. Two of these blocks included 6 transects with 2 each from 3 of the treatments. In these blocks, the older reclaimed treatment was not represented because reclaimed land of this age was not present in close proximity to the other 3 treatments. Another 2 blocks included 8 transects with 2 from each of the 4 treatments. The fifth block included 10 transects: 2 from each of the 4 treatments plus an additional 2 transects in an older reclaimed area that is now dominated by pine woodlands. Transects within each block were trapped concurrently, thus minimizing temporal effects on comparisons between treatments. Blocks contained transects located as close to one another as the landscape allowed to minimize spatial differences. The traps were rotated weekly to a new block until each block was trapped 2 times over the summer. Traps were pre-baited for 1 night and then opened and checked for 3 consecutive nights. The period between trapping sessions at a given block was about 25 days.

Captured animals were identified, weighed, sexed, and examined for reproductive status. All individuals except members of the shrew family (Soricidae) were marked with numbered metal ear tags before release. Because shrews have small external ears, these species (short-tailed shrew and masked shrew (*Sorex cinereus*) were marked by toe-clipping (ACUC# 9904-10). Any individuals that died in traps were saved as voucher specimens.

Statistical methods included calculations of relative abundance of small mammals, expressed as the number of individuals trapped per 100 trap nights, with recaptures excluded. A correction was made for sprung traps in calculations of trap effort; one-half a trap night was subtracted for each trap sprung for any reason, including the capture of an animal (Nelson and Clark 1972, Beauvais and Buskirk 1999). Species richness was calculated as the number of species captured per transect. A randomized block analysis of variance (ANOVA) (Zar 1999) was used to compare total relative abundance, species-specific relative abundance, and species richness among treatments. Concurrently trapped transects were considered blocks for this model since temporal and spatial factors were minimized by the design. When differences between treatments were detected by the ANOVA, Duncan's multiple comparison test was used to find where the differences occurred. Statistical tests were considered significant at $\underline{P} \leq 0.05$.

Surveys were not conducted for larger mammals such as carnivores and ungulates (Order Carnivora, Order Artiodactyla); however, any incidental sighting was recorded to document their presence on the study area. Surveys also were not conducted for bats (Order Chiroptera), though an important part of the mammalian fauna, due to time and logistical limitations. Because small mammal trapping initially began in 1999, we chose to continue sampling this group in 2000.

Herpetofaunal Abundance

Pitfall and funnel traps, when associated with drift fence arrays, are extremely effective in collecting large numbers of herpetofauna and in capturing the majority of species from a given area with minimal effort (Campbell and Christman 1982, Vogt and Hine 1982, Jones 1986, Bury and Corn 1987, Mengak and Guynn 1987, Pais et al. 1988, Corn 1994). Campbell and Christman (1982) also found that drift fence arrays can be used to "…provide a clear indication of relative abundances between habitat types." Drift fence arrays have been used effectively in both forested areas (Bury and Corn 1987) and grassland/wetland areas (Vogt and Hine 1982, Homyack 1999). Accordingly, we chose this method to gain relative abundance and species richness data for comparison among the 4 treatments.

Because of their ability to intercept animals traveling in any direction, we used plus (+) shaped arrays with 15 m of central separation (Fig. 16; Campbell and Christman 1982, Corn 1994). Fifteen meter sections of 30-cm tall plastic silt fencing, supported by wooden stakes, were used to construct the drift fence (Enge 1997). Silt fencing is lighter and cheaper than the traditionally used aluminum flashing, but is durable and appears to work just as well (Enge 1997, Homyack 1999). An 18.9-L plastic bucket (pitfall trap), was buried flush with the surface at the end of each individual drift fence (Campbell and Christman 1982, Vogt and Hine 1982, Pais et al.

1988, Corn 1994). Plastic bucket lids, elevated by sections of untreated 2 x 4, served as shade covers when the traps were open and were inverted to close traps when necessary (Homyack 1999). To prevent desiccation of captured herpetofauna, 2-3 cm of water was placed in the bottom of each trap (Vogt and Hine 1982). In addition, the water kills any inadvertently captured small mammals or arthropods that may otherwise injure trapped herpetofauna (Vogt and Hine 1982). All drift fence segments had funnel traps (minnow trap #1275, Frabill, Jackson, Wisc.) located at the midpoint on either side of the fence (Campbell and Christman 1982, Vogt and Hine 1982, Bury and Corn 1987, Pais et al. 1988, Corn 1994). Soil or leaves were brushed into the entrance of funnel traps to create a more natural entrance for herpetofauna (Campbell and Christman 1982). Sections of silt fencing were attached to funnel traps to provide shade for captured organisms (Homyack 1999). The 4 arms of the 'plus' and associated traps made up the drift fence array.

Arrays overlapped 12 randomly selected songbird point count stations that were positioned 35 m from a stream or rip-rap channel (Fig. 12). Arrays were distributed over the 3 watersheds with 3 arrays per treatment (Table 2). All arrays were opened simultaneously for 5 days in March and 8-12 consecutive days during each month of the field season (March – September 2000). While traps were open, they were visited at least every other day (Campbell and Christian 1982, Vogt and Hine 1982, Corn 1994). Captured organisms were identified to species using field guides, marked so that individuals recaptured during a trapping session could be identified, and released 3 m from the drift fence array (Campbell and Christian 1982, Vogt and Hine 1982, Fellers et al. 1994). Frogs, toads, salamanders, and lizards were marked using toe clipping where each individual was given a unique number based on its toe clips. When possible, missing or deformed toes were used to identify an individual rather than clipping a toe. Snakes initially were marked with a v-shaped notch at the edge of a ventral scale. We later marked snakes by painting a number on the back with white-out. We also recorded the trap number and trap type (Fig. 16) for each individual captured. Voucher specimens of all unusual or hard-to-identify herpetofauna were killed and preserved according to the techniques described by McDiarmid (1994b). Small mammals were identified to species and, if they were alive, released.

Because length of the trapping periods varied somewhat, the number of animals captured in all pitfall and funnel traps on each array during a trapping period were summed and divided by the number of nights the traps were open in a trapping period (Corn 1994). These values (mean captures per array-night in each trapping period) were used in statistical analyses. Although few individuals were recaptured, recaptures were excluded from data summaries. Treatments were compared with ANOVA with mean abundance and richness as the dependent variables and treatment, trapping period, and the interaction between treatment and trapping period as independent variables.

Quality Control Procedures

Sampling was conducted on 3 (Mud River, Spruce Fork, and Twentymile Creek) of the 5 watersheds chosen by the EPA. The Island Creek and Clear Fork sites were not selected because past and existing land use would confound study results. Four treatments (intact forest; fragmented forest; young reclaimed mine: grassland; and older reclaimed mine: shrub/pole stage) were replicated at each site (Tables 1 and 2). An unbalanced sampling design among treatments and taxa was necessary because of logistics (e.g. point counts required less time to sample per point than do small mammal transects) and a lack of some treatments at some sites. Multiple replicates allowed us to incorporate variation across sites, and enabled us to make statistical inferences regarding species abundances and diversity among treatments. Sampling points (i.e., point counts, transect lines, and trap arrays) were distributed to be representative and to minimize spatial differences, while at the same time maintaining sampling efficiency. Concurrent sampling among taxa and sites was used to minimize temporal effects.

Quality control was insured through a hierarchical oversight procedure. Data on each taxon was collected by a 2-3 person team. Each team included a supervisor (MS students for mammal and raptor studies, trained technician for herpetofaunal study, and PhD research biologist for songbird studies) and field technicians. Overall data collection was supervised by the PhD research biologist in coordination with project PIs. This team approach allowed for consistent data collection during the 1999 and 2000 field seasons. Individual team supervisors remained the same in both years, while field technicians changed the second year. This approach insured precision and consistency in methodologies and reduced sampling error.

Data collection adhered to established protocols (e.g. point counts, trapping, drift fences, raptor surveys) for each taxon and are detailed in the methods. Technicians received ample training in methodologies and species identification (e.g. simultaneous point counts) prior to any unsupervised data collection. Voucher specimens of unusual or hard-to-identify mammalian or herpetofaunal species were collected and preserved to insure data accuracy.

Results and Discussion

Habitat at Sampling Points

Habitat variables were measured at all sampling points in 1999 and 2000 (Table 6). Nineteen variables were measured in all treatments. Means for all habitat variables by treatment and mine are found in Appendix 4

Stem densities of saplings, poles, and trees in 5 size classes all differed significantly among treatments (Table 7). Pole density, and densities of trees >8-23 cm and >23-38 cm were higher in fragmented and intact forest than in the grassland and shrub/pole treatments and also higher in the shrub/pole treatment than in the grassland treatment. Density of trees >53-68 cm was greater in fragmented forest than in the intact forest, grassland, and shrub/pole treatments, and greater in the intact forest treatment than in the grassland and shrub/pole treatments. Trees >68 cm were more abundant in the intact forest and fragmented forest treatments than in the grassland and shrub/pole treatments (Table 7).

Statistical analysis revealed treatment by mine interactions for saplings and trees >38-53 cm (Table 7); therefore treatments were compared on individual mines, and mines were compared in individual treatments. Sapling density was higher at the Hobet and Daltex mines than at the Cannelton mine in the grassland treatment, and trees >38-53 cm had higher density in the shrub/pole treatment on the Cannelton mine than the Hobet mine and higher density in the intact treatment at the Daltex and Hobet mines than the Cannelton mine (Table 8). At all 3 mines, sapling density was higher in the shrub/pole, fragmented forest, and intact forest treatments than in the grassland treatment. At the Cannelton mine density of trees >38-53 cm differed among all 4 treatments, with the highest density in the fragmented forest treatment and lowest density in the grassland treatment (Table 9). At the Hobet mine, density of trees >38-53 cm was higher in both fragmented and intact forest treatments than in grassland and shrub/pole treatments (Table 9).

Ground cover variables differed significantly among treatments. Although water cover was highest in the fragmented forest treatment than in the other 3 treatments and higher in the intact forest treatment than in the grassland or shrub/pole treatment (Table 7), cover of standing water averaged <1.2%. Woody debris and moss cover were higher in fragmented and intact forest than in the grassland and shrub/pole treatments. Green cover was higher in the shrub/pole treatment than in the other 3 treatments, and higher in the grassland treatment than in the fragmented forest or intact forest treatments (Table 7).

Bareground cover and litter cover had significant treatment by mine interactions. Bareground cover was higher at the Cannelton mine in the fragmented forest treatment than at the other 2 mines and higher at the Daltex mine than the Hobet mine in the grassland treatment (Table 8). Litter cover was higher at the Hobet mine than the other 2 mines and higher at the Daltex mine than the Cannelton mine in the grassland treatment (Table 8). Bareground and litter cover also differed among treatments at the Cannelton and Hobet mines. At the Cannelton mine litter cover was higher in the fragmented and intact forest treatments than the shrub/pole and grassland treatments, and higher in the shrub/pole treatment than in the grassland treatment (Table 9). At the Hobet mine, litter cover differed among all treatments; it was highest in the fragmented forest treatment, followed by intact forest, grassland, and shrub/pole treatments (Table 9). Bareground cover at the Cannelton mine was higher in the fragmented forest, intact forest, and grassland treatment than in the shrub/pole treatment. At the Hobet mine, bareground cover was higher in the fragmented forest treatment than in the shrub/pole treatment, and higher in the intact forest treatment than in the shrub/pole and grassland treatments (Table 9).

Slope, aspect code, elevation, and distances to nearest minor, habitat, and mine/forest edges also were compared among all 4 treatments (Table 7). Distance to nearest minor edge was greater in the grassland treatment than in the other 3 treatments (Tables 6-7). There were significant mine x treatment interactions for slope, aspect code, elevation, distance to closest habitat edge, and distance to nearest mine/forest edge. The differences among treatments and mines for these variables are found in Tables 8-9.

Six variables were compared between grassland and shrub/pole treatments and mines. Litter depth was higher on the Hobet mine than the Cannelton and Daltex mines and higher in the Daltex mine than the Cannelton mine (Table 7). The Robel pole index was higher on the Cannelton mine than the other two mines and higher on the Daltex mine than the Hobet mine (Table 7). Forb cover was higher on the Cannelton and Daltex mines than on the Hobet mine (Table 7). The other variables all showed significant treatment by mine interactions. Grass height was higher at the Hobet mine than at the Daltex and Cannelton mines in the grassland treatment and higher at the Hobet mine than the Cannelton mine in the shrub/pole treatment (Table 9). Ground cover of grass and shrubs differed among mines, but not between grassland and shrub/pole treatments (Table 8-9).

Canopy height, percent canopy cover in 6 layer classes, and the structural diversity index were compared among the fragmented forest, intact forest, and shrub/pole treatments (Table 7). Percent canopy cover in 5 layer classes differed among treatments but not among mines (Table 7). There were treatment by mine interactions for canopy height and cover from >3-6 m. Canopy height was higher at the Cannelton mine than the Daltex and Hobet mines in the fragmented forest treatment, and was higher at the Daltex mine than the Hobet mine in the intact treatment (Table 8). Canopy cover from >3-6 m was higher at the Cannelton and Daltex mines than the Hobet mine in the intact forest treatment (Table 8). This cover layer also differed among treatments at the Cannelton and Hobet mines (Table 9). It was higher in the fragmented and intact forest treatments than the shrub/pole treatment at the Cannelton mine. At the Hobet mine it was highest in the intact forest, followed by fragmented forest and shrub/pole treatments (Table 9).

The majority of minor edge types in the grassland treatment were open-canopy roads and valleyfills (Table 10). In the shrub/pole treatment the majority of minor edges also were open-canopy roads and valleyfills. The majority of minor edge types were stream and open-canopy road in fragmented forest, and partially-open canopy road and stream in intact forest (Table 10). These percentages are based on subplots and not point count centers, because subplots in a point count circle could occur closer to different edge types. The average distances to any edge type were 110 m in grasslands, 67 m in shrub/pole, 38 m in fragmented forest, and 66 m in intact forest. Again, these averages are based on subplots and not the point count center.

Fifteen tree/shrub species were observed on grassland sampling points, with predominant species including autumn olive, European black alder, blackberry/raspberry, multiflora rose, red maple, sourwood, and white pine (Appendix 2). In the shrub/pole treatment, 38 species were observed, with black locust being the most predominant. Twenty-seven species were observed on the Cannelton mine in shrub/pole habitat, and twenty-one species were observed on the Hobet mine site. An additional 7 species were observed in shrub/pole treatment at the Hill Fork site, which was a valley fill associated with a contour mine. Sixty-three species were observed in fragmented forest, and 60 species were observed in intact forest (Appendix 2).

Songbirds

Comparison of Expected to Observed Bird Species

Buckelew and Hall (1994) in *The West Virginia Breeding Bird Atlas* (WV BBA) identified 92 bird species as being either “probable” or “confirmed” breeders in the counties of Boone, Fayette, Kanawha, and Logan in southern West Virginia (Table 11). Only 8 of these species were not observed during the course of this study based on pond surveys, point count surveys and incidental observations: House Wren, Warbling Vireo, Pine Warbler, Winter Wren, House Sparrow, Purple Martin, House Finch, and Rock Dove. These 8 species are found in habitats that were not surveyed during this study. The House Wren and Warbling Vireo are found in bottomland hardwood thickets and around human habitations, and the Pine Warbler, as its name suggests, is restricted to stands of pines. The House Sparrow, House Finch, Rock Dove, and Purple Martin also are found around human dwellings and generally are not often observed in the types of habitat that we surveyed. The Winter Wren is most often observed in higher elevations in West Virginia, and it is likely that this species occurs in the higher elevations of eastern Fayette County. Our study site (Cannelton mine) was located in southwestern Fayette County.

Several grassland and shrub species that we observed on mine sites were not listed by the WV BBA as being probable or confirmed breeders in southern West Virginia (Table 11). These included: Bobolink, Dickcissel, Grasshopper Sparrow, Henslow’s Sparrow, Horned Lark, Ring-necked Pheasant, Vesper Sparrow, Willow Flycatcher, Blue Grosbeak, and Purple Finch. Dickcissels and Horned Larks historically were midwestern species that have moved east from the prairies (Askins 1999). We observed several male Dickcissels defending territories and 1 female carrying food in Logan County at the Daltex mine; it is probable that this species is breeding there. They were only observed incidentally in Boone, Fayette, and Kanawha counties at the Hobet and Cannelton mines. Two Horned Lark nests were found, 1 in Boone County at the Hobet 21 mine and 1 in Logan County on the Daltex mine. Grasshopper Sparrows, a species listed as "rare" by the West Virginia Wildlife and Natural Heritage Program (2000), were abundant on our grassland sites. We found several nests of Grasshopper Sparrows at all 3 mine sites, and thus, this species is a confirmed breeder in these areas. One nest of a Willow Flycatcher was found by observers working on the Cannelton mine (D. Stover, personal communication). Willow Flycatchers and Blue Grosbeaks were most often observed defending territories in blocks of autumn olive. Several female Blue Grosbeaks were observed during the study, but no nests were found. Only 1 male Purple Finch was observed, at the Cannelton Mine, and it was likely just an incidental occurrence. Ring-necked Pheasants were observed at the Hobet mine, but it is suspected that these are released birds and not wild birds. No females or nests were located for this species.

Typical grassland species that were rare or absent on our sites included Henslow's Sparrow, Savannah Sparrow, Vesper Sparrow, and Bobolink. Henslow's Sparrow and Vesper Sparrow were only recorded at the Logan County mine in very low densities, and no females were observed, so it is likely that neither species are breeding at our mine sites. Henslow's Sparrow populations are rare, scattered, and local in distribution (Herkert and Glass 1990) and are listed as a "rare" species in West Virginia (West Virginia Wildlife and Natural Heritage Program 2000). They prefer grasslands with tall, dense vegetation with a well-developed litter layer (Herkert and Glass 1990). Due to the young age of our sites, the habitat may not be suitable for this species. Vesper Sparrows prefer grasslands with high amounts of bareground for nesting (Strait 1981), courtship, and foraging (Wray 1982). Strait (1981) found that Vesper Sparrows prefer to nest in areas with a mean bareground cover of 29%, and Wray (1982) found that bareground cover on Vesper Sparrow territories averaged 35.5%. Our grassland study sites only had a mean bareground cover of 7.7%, which may have limited this species on our sites. Bobolinks, also listed as a "rare" species in the state (West Virginia Wildlife and Natural Heritage Program 2000), were only observed early in the spring and were assumed to be migrating. Savannah Sparrows were not observed on any of our sites, although they are a common grassland species in other areas of West Virginia (Wray et al. 1982, Warren and Anderson, unpub. data).

Historically, grassland bird species in the eastern United States were restricted to limited patches of habitat interspersed among forest stands (DeSelm and Murdock 1993). Virtually no natural grasslands are believed to be have been present historically in the Allegheny and Cumberland Plateaus of West Virginia (DeSelm and Murdock 1993), where most MTMVF occurs in this state. Native grasslands in these physiographic provinces are primarily found in the moderately deep to shallow soil of uplands (DeSelm and Murdock 1993). Grassy balds composed of moonshine grass (*Danthonia compressa*) with scattered hawthorn trees (*Crataegus* spp.) occur on high elevation mountain tops in the Allegheny Mountain and Ridge and Valley provinces of West Virginia. Heath barrens of heath shrubs and low-growing plants, as well as glades similar to bog communities, also occur in these provinces (Strausbaugh and Core 1977). Although natural grasslands were limited, grasslands created by Native Americans for agriculture and hunting did exist (Askins 1999). Presently, human-made grasslands in these provinces include pastures, old fields, lawns, golf courses, and surface mines. Grassland birds typically observed in these habitats include Horned Lark and Dickcissel, that have moved east from the midwestern prairies, and species such as the Eastern Meadowlark, Bobolink, Savannah Sparrow, and Grasshopper Sparrow, that are assumed to have expanded into these areas from coastal and marsh grasslands (DeSelm and Murdock 1993, Askins 1999). All of these species were reported by early ornithologists in the East (Askins 1999).

Several wetland species not listed by the WV BBA were observed at pond sites on reclaimed mines (Table 11). Fifty-seven species were observed during pond surveys within 50 m of ponds on MTMVF areas (Table 11). The majority of these species were grassland and edge species that were detected in habitats adjacent to ponds. Ducks, geese, wading birds, and shorebirds all used the ponds. Mallards and Canada Geese were observed frequently, as well as Green and Great Blue Herons. During migration several shorebirds were observed using the ponds, including Greater and Lesser Yellowlegs, and Spotted and Solitary Sandpipers. Three species of swallows (Barn, Northern Rough-winged, and Tree) as well as Chimney Swifts were observed foraging over ponds, whereas Cliff Swallows were observed foraging in adjacent grassland habitat. Sandpiper species and yellowlegs were likely migrating during the May pond surveys. None of these species were observed during the July pond surveys. Many of the species we observed also have been documented by other researchers examining wetlands on surface mines (Allaire 1979, Perkins and Lawrence 1985, Brooks et al. 1985, Krause et al. 1985, Lawrence et al. 1985, McConnell and Samuel 1985).

The West Virginia Gap Analysis Lab (J. Straiger, pers. comm.) also provided us with a list of species expected to occur in southern West Virginia based on remote sensing data of the available habitat (Table 11). Most of the species predicted to occur in our areas were observed during this study. A few exceptions included Chestnut-sided Warbler, Rose-breasted Grosbeak, Black-throated Blue Warbler, Canada Warbler, and Winter Wren. All of these species are associated with the northern hardwood forest type (Hinkle et al. 1993) and typically occur at high elevations (>900 m) in the Allegheny Mountains of West Virginia (Wood et al. 1998, Demeo 1999, Weakland 2000). This habitat and elevation were absent in our study area, and thus it is not surprising that we did not observe these species. Wetland species that Gap predicted to occur that we did not observe included the American Black Duck, Hooded Merganser, and Swamp Sparrow. We observed all of the grassland species that they predicted as well as all of the edge species, except for the Chestnut-sided Warbler, mentioned above, and the Warbling Vireo, which is found in bottomland hardwood thickets and near human dwellings.

Songbird Abundances in Grassland and Shrub/pole Habitats

We observed 63 species of birds in reclaimed sites with 30 species in the grassland treatment and 41 species in the shrub/pole treatment on MTMVF areas in southern West Virginia during point count surveys (Table 12). The most abundant songbird species in grassland areas of reclaimed mines were Grasshopper Sparrow, Eastern Meadowlark, Red-winged Blackbird, Horned Lark, and Dickcissel. Species associated with shrub/pole habitat also were observed using small shrubs as perches and nesting in blocks of autumn olive at our grassland points. These species included Indigo Bunting, Common Yellowthroat, Willow Flycatcher, Song Sparrow, American Goldfinch, Blue Grosbeak, Brown Thrasher, Orchard Oriole, Field Sparrow, and Yellow-breasted Chat. The average abundances of bird species by mine and treatment are found in Appendix 3.

The most abundant species in the older reclaimed areas (shrub/pole habitats) included American Goldfinch, Blue-winged Warbler, Common Yellowthroat, Eastern Towhee, Field Sparrow, Indigo Bunting, Northern Cardinal, Prairie Warbler, White-eyed Vireo, Yellow Warbler, and Yellow-Breasted Chat (Table 12). This bird community included all 4 habitat guilds because these areas had a mixture of vegetation characteristics (grass/forb, shrubs, and trees of small and moderate size).

Point counts measure relative abundance, so to compare our results with other studies we converted our abundance estimates to density estimates by dividing the mean number of birds observed by the number of hectares (0.79) in a 50-m radius point count circle. However, it was difficult to compare grassland bird densities with other studies because of differences in methods. For example, spot mapping and territory flush methods primarily count singing males or male territories in a defined area, whereas point counts and strip transects record all birds either seen or heard, including females and juveniles. Thus, our estimates may be higher than those observed in studies that used territory count methods.

Densities of Grasshopper Sparrows, our most abundant species in the grassland treatment, were much higher than those reported in other studies (Table 13). Allaire (1979) found a much lower density on 1-4-yr old reclaimed MTMVF areas in eastern Kentucky. Our sites have been reclaimed for at least 5 years, and the average age was 11 years. Thus, Grasshopper Sparrows may have had more time to settle on our sites than Allaire's (1979). Additionally, vegetative structure on our mines may have been more suitable for Grasshopper Sparrows than the vegetation on his sites. LeClerc (1982) found Grasshopper Sparrows preferred mines with a high amount of forb cover and a low amount of bare ground cover. Our sites were more developed vegetatively than Allaire's (1979). The amount of bareground cover on his sites averaged 17%, whereas ours averaged only 8%, and the height of foliage on his sites averaged 6.4 dm, wheareas ours averaged 7.3 dm. Other studies on reclaimed surface mines and in other types of grassland habitat report lower densities of Grasshopper Sparrows (Table 13), but these differences may be due to the method used to calculate density. Territory mapping and flushes estimate the number of territory-holding males in an area while point counts include all singing males. Our study sites may have contained high numbers of unmated males (also see nest success section below). The higher numbers detected in our study were not due to overall population increases since Allaire's study. Breeding Bird Survey data indicate a declining trend in grasshopper sparrow populations in the 2 physiographic provinces (Cumberland Plateau and Ohio Hills) that overlap our study sites (Sauer et al. 2000).

With the exception of Bobolinks and Savannah and Vesper Sparrows, densities of other species on our sites fell within the ranges reported by other researchers on reclaimed mines and other grassland habitat (Table 13). Neither Savannah nor Vesper Sparrows were observed in 2000 on our sites, and only 2 Vesper Sparrows were heard in 1999 at the Logan County mine. Bobolinks were only observed on 2 point counts in 2000, and they may have been migrants. Our sites lie at the southern extreme of the breeding range for these 3 species (Buckelew and Hall 1994).

Songbird abundances in our shrub/pole community are similar to those found by others who have examined surface mines (Brewer 1958, Chapman 1977, Crawford, et al. 1978, LeClerc 1982, Wray 1982). Because our shrub/pole treatment included a few sites on the oldest MTMVF area in West Virginia (~26 years) compared to an average of 18 years (range 13-25) for the remaining sites, we examined these different-aged sites separately (Table 14). Overall species richness and total abundance were similar between younger and older shrub/pole areas with a 65% similarity in the bird community (Table 14). Our results were similar to abundances reported by Denmon (1998) on early successional sites (33% reclaimed mines, the remainder on unmined lands) throughout West Virginia (Table 14). In addition, all of the species listed by Hinkle et al. (1993) as being present in shrub habitat or shrub-small tree habitat in the mixed mesophytic forest region were present on our shrub point counts. One shrub/pole species of conservation interest is the Golden-winged Warbler, which is listed by Partners in Flight as a species of concern in the entire Northeast region. We only observed this species at the Cannelton mine site at 3 point count stations, and it is possible that the Hobet and Daltex mine sites were out of this species' elevational or geographic ranges. If this species is limited by range, it is unlikely that MTMVF will increase habitat for this species in the Mud River and Spruce Fork watersheds.

Songbird Abundances in Fragmented and Intact Forest

Mixed mesophytic forests support the richest and most abundant avifaunal community in the eastern United States outside of bottomland and swamp habitats (Hinkle et al. 1993). All of the bird species listed by Hinkle et al. (1993) as being present in mature, mixed mesophytic forest were observed on our sites. We observed 50 species of birds in forested sites with 47 species in the fragmented forest treatment and 43 species in the intact forest treatment during point count surveys (Table 12). The most abundant forest interior species on our sites included Acadian Flycatcher, Blue-headed Vireo, Cerulean Warbler, Kentucky Warbler, Ovenbird, and Wood Thrush (Table 12).

Songbird abundances in our intact forest sites generally were similar to those reported by other researchers in undisturbed forests of the mixed mesophytic forest region (Anderson and Shugart 1974, Allaire 1979, Wood et al. 1998, Demeo 1999; Table 15). Two species of note, however, are Ovenbird and Cerulean Warbler. Ovenbirds occurred at higher densities on our intact treatment than in any other study (Table 15). The Cerulean Warbler, a species of high concern in the eastern United States, occurred at higher densities on our sites than in other areas of West Virginia, though at lower densities than in Kentucky. They were observed at 40% of all intact forest point counts and at 28% of fragmented forest point counts. Cerulean Warblers have been declining in many parts of their range, and southwestern West Virginia may represent a significant source population for this species in the eastern United States (Rosenberg and Wells 1999). It is estimated that 47% of the Cerulean Warbler population in North America occurs in the Ohio Hills physiographic area (Rosenberg 2000), which includes part of our study area.

Abundances of several species of songbirds on our study sites differed between fragmented forest and intact forest (Table 12). Six species were significantly more abundant in intact forests: Acadian Flycatcher, Ovenbird, American Redstart, Hooded Warbler, and Brown-headed Cowbird in both 1999 and 2000, and the Scarlet Tanager in 1999 (Table 12). Red-eyed Vireos and Indigo Buntings were significantly more abundant in fragmented forest than intact forest in both years, while 6 species (American Goldfinch, Downy Woodpecker, Louisiana Waterthrush, Northern Parula, Pileated Woodpecker, and Yellow-billed Cuckoo) were more abundant during 1 year. The Louisiana Waterthrush occurs near streams, where it nests in stream banks and forages in the stream. Proportionally more of the fragmented forest sampling points were located along streams than in the intact forest treatment. Therefore, we ran a subsequent analysis for this species using only points located within 50-m of a stream. With this restriction we found no significant differences in abundance of this species between fragmented and intact treatments ($F=0.36$, $P=0.55$). The American Goldfinch and Indigo Bunting are edge species, while the Downy Woodpecker, Northern Parula, Red-eyed Vireo and Yellow-billed Cuckoo are considered interior-edge species. These birds may be responding to the higher amount of edge in fragmented forest than in intact forest (Temple 1986).

The Brown-headed Cowbird had very low abundance in our study (0.07 birds/count). This species was observed only at 1 intact forest point count in 1999, and only at 1 fragmented forest and 7 intact forest point counts in 2000. The species was not observed in the Twentymile Creek watershed. Thus, we suspect that Brown-headed Cowbird parastism is likely to be low in this region and not a significant cause of nest losses. The abundance of cowbirds is relatively low in other parts of West Virginia as well (Demeo 1999, Weakland 2000).

High moisture availability in mature mixed mesophytic forests may contribute to the high densities of many species of songbirds in these habitats as compared to forests with lower ambient moisture, such as xeric oak-hickory forests (Hinkle et al. 1993). Species that are abundant and common in mixed mesophytic forests, such as Cerulean Warblers, Kentucky Warblers, Acadian Flycatchers, and Ovenbirds, are frequently less abundant and rare in drier forests (Hinkle et al. 1993). Several species in our study had higher abundance in intact forest than fragmented forest. It is possible that fragmented stands are drier because the microclimate has been altered (Faaborg et al. 1995) and that songbirds are responding negatively to this change. In addition, fragmentation also may negatively affect songbird species by leading to higher rates of predation, cowbird parasitism, interspecific competition, and to lower pairing success and nesting success (Faaborg et al. 1995). Additionally, some species have “minimum area requirements” and are not found in fragments below a certain size threshold. As forest size is reduced, specific microhabitats upon which some species depend also may be reduced or even disappear. Consequently, species associated with those microhabitats may disappear or decline in fragmented forest (Faaborg et al. 1995). The Ovenbird, Acadian Flycatcher, Hooded Warbler, and American Redstart, species that were more abundant in intact forests than fragments in our study, prefer large blocks of mature forest in eastern deciduous forests (Robbins 1980, Blake and Karr 1987). The Ovenbird is known to have lower pairing success and lower nest survival in forest fragments than in intact forests (Gibbs and Faaborg 1990, Robinson et al. 1995, Hagan et al. 1996), and the Hooded Warbler also has lower nest survival in fragmented landscapes (Robinson et al. 1995).

Species-specific Logistic Regression Models

The presence/absence of 10 forest-dwelling songbird species of conservation priority for the region were related to specific habitat variables. Logistic regression models were fit for each species and none were rejected due to lack-of-fit (Hosmer and Lemeshow goodness-of-fit tests, $P>0.10$),

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<u>Cerulean Warbler</u>
The Cerulean Warbler, with the highest conservation priority rating (Table 5), was found to be positively related to percent slope and percent canopy cover from >6-12 m (Table 16). The Ohio Hills and Northern Cumberland Plateau physiographic provinces where MTMVF mining is prominent are within the core area for the Cerulean Warbler. It is estimated that 46.8% of this species’ population is found within the Ohio Hills province alone (Rosenburg 2000). This species prefers large tracts of mature forests with large, tall trees (P. Hamel, unpub. rept.). We found Ceruleans more often on steeper slopes, as did Dettmers and Bart (1999) in southeastern Ohio. Based on habitat preferences, it is reasonable to conclude that continued

MTMVF mining will negatively impact Cerulean Warbler abundance in southwestern West Virginia.

Lousiana Waterthrush

The Lousiana Waterthrush, with the second highest conservation rating, was negatively related to percent bareground cover and pole density, and was positively related to percent moss cover (Table 16). This species is found in large tracts of mature forest and nests on the ground along stream banks (Whitcomb et al. 1981, Ehrlich et al. 1988). Bushman and Therres (1988) suggested that wooded streambanks and ravines be protected in order to maintain this species. Given valleys and streams are covered by MTMVF operations and reduces mature forest cover, it is logical to conclude that this species also will be negatively affected by loss of streamside forest habitat from this type of mining.

Worm-eating Warbler

This species was positively related to percent woody debris cover and negatively related to percent canopy cover from >12-18 m, aspect, percent litter cover, and elevation (Table 17). Worm-eating Warblers typically are found in ravines and on hillsides in deciduous woods where they nest on the ground in leaf litter (Ehrlich et al. 1988, Dettmers and Bart 1999). They are most abundant in mature forests, although they may be found in young- and medium-aged forest stands as well (Bushman and Therres 1988). Robbins (1980) and Whitcomb et al. (1981) suggested that this species requires large tracts of mature forest and may have a low tolerance for fragmentation. The greatest threat to this species from MTMVF is the loss and fragmentation of forested habitat.

Kentucky Warbler

Kentucky Warblers were present at points with a high percent of canopy cover from >6-12 m, and low sapling and pole density and also were present more often at lower elevations (Table 17). Kentucky Warblers prefer rich, moist forests and bottomlands with well-developed ground cover (Bushman and Therres 1984). This species appears to be moderately affected by fragmentation and may be found in small woodlots, but in Maryland the highest frequency of occurrence for this species was in forests from 130-700 ha in size (Bushman and Therres 1988). Loss of wooded ravines and bottomlands could negatively affect this species.

Acadian Flycatcher

This species was one of our most abundant birds and abundance was correlated to many habitat variables (Table 18). It was positively related to trees >68 cm, and negatively related to saplings and trees 8-23 cm dbh, indicating an association with mature forests. It also was positively related to distance from mine/forest edge, structural diversity, and percent bareground, and negatively associated with elevation. Acadian Flycatchers prefer moist ravines and stream bottoms. Dettmers and Bart (1999) considered this species to be a habitat "specialist" at the microhabitat (i.e. territory or home range) level. Bushman and Therres (1988) found that Acadian flycatchers prefer forests with high canopy cover, large trees, and an open understory. This species prefers large blocks of mature contiguous forest for breeding, and appears to avoid edges. We found this species to be more abundant as distance from mine edge increased and more abundant in intact forest, which could indicate that MTMVF mining is detrimental to this species.

Wood Thrush

Wood Thrush were positively related to density of trees >23-38 cm dbh and negatively associated with elevation (Table 18). Wood Thrush are found in deciduous and mixed coniferous-deciduous forest, with highest densities occurring in the Appalachian Mountain region (James et al. 1984). They prefer mature forests with some small trees in the understory for nesting and a moist, leafy litter layer for foraging (James et al. 1984).

Yellow-throated Vireo
Presence of this species was related to several variables. It was positively related to percent canopy cover from 6-12 m, aspect, slope, elevation, and density of trees from 38-53 cm (Table 19). It was negatively associated with distance to mine/forest edge and percent bareground. It is most abundant in mature forests and appears to prefer stream borders and bottomland forests (Bushman and Therres 1988). Yellow-throated Vireos appear to have a low tolerance for forest fragmentation (Whitcomb et al. 1981). MTMVF mining could potentially reduce abundance of in this species because of its preference for mature forest along streams, which may be lost due to mining.

Hooded Warbler
Hooded Warblers were positively related to percent cover of woody debris and pole density (Table 19). Hooded Warblers typically are found in moist deciduous forests and ravines with a well-developed understory (Ehrlich et al. 1988), but also may be found along ridges with a high density of shrub stems (Dettmers and Bart 1999). It is suspected that this species is fragmentation-sensitive (Bushman and Therres 1988), and we found it to occur at higher abundances in intact than fragmented forest sites.

Scarlet Tanager
This species was negatively associated with percent bareground cover. They were positively associated with elevation, percent slope, density of trees from >38-53 cm, and canopy cover from >12-18 m (Table 20). This species may be found in a wide range of successional stages of forests, but is most abundant in mature woods with a dense canopy (Bushman and Therres 1988). This species does not appear to be as fragmentation-sensitive as other forest interior species, and may tolerate smaller forests and edges (Bushman and Therres 1988); however, it was more abundant in our intact than fragmented forest sites during 1 year of the study., and was more common at points further away from mine/forest edge.

Black-and-white Warbler
Black-and-white Warblers were positively associated with pole density, percent ground cover of moss, aspect, and distance from mine/forest edge (Table 20). It was negatively associated with percent canopy cover from 3-6m and sapling density. This species nests on the ground in deciduous and mixed forests (Ehrlich et al. 1988). It appears to prefer pole-stage stands (Bushman and Therres 1988), but it is fragmentation-sensitive and was not found breeding in forests <70 ha in size in Maryland (Whitcomb et al. 1981).

Yellow-billed Cuckoo
The Yellow-billed Cuckoo was positively related to percent cover of woody debris (X^2=3.99, P=0.05) and negatively associated with elevation (X^2=7.00, P=0.01) and aspect ((X^2=2.99, P=0.08). This species is a PIF priority species for the region (Rosenberg 2000), but we observed it at only 9 sampling points in the 2 years of the study. Less than 1% of the population occurs in this region (Rosenberg and Wells 1999), and MTMVF is not likely to severely impact the population as a whole.

Other Species

The Swainson's Warbler, a species of concern in the region and a rare species in West Virginia (West Virginia Wildlife and Natural Heritage Program 2000), is typically, in West Virginia, found only in areas of dense rhododendron (Buckelew and Hall 1994). We observed this species in the Twentymile Creek watershed along Hughes Fork. Further MTMVF in this watershed could impact this species, but the effect on the population as a whole will be minimal, since <2% of the population is found in the Ohio Hills province and West Virginia is on the periphery of its range (Table 5) . The Eastern Wood-pewee is a species of conservation priority (Action level III) in the region, but we only observed it at 1.2% of our forested point counts. The Black-billed Cuckoo is a PIF priority species for this region (Rosenberg 2000), but it appears to be relatively rare; it was only observed incidentally in early successional habitat during this study and was not detected during point count surveys.

Comparison of Guild Abundances Among Treatments

All of the habitat guilds differed significantly among treatments (Table 21). As expected, the grassland guild was more abundant in the grassland treatment than in shrub/pole, fragmented forest, or intact treatments. Edge species also followed a typical pattern: they were most abundant in shrub habitat, followed by grasslands, then by fragmented and intact forest (Table 21). Interior-edge species were most abundant in the fragmented and intact forest treatments, followed by the shrub/pole and grassland treatments. Forest interior species were more abundant in intact forest, followed by fragmented forest, shrub/pole, and grassland treatments. Significantly higher abundance of forest interior species in intact than fragmented forests suggests that this group is negatively affected by habitat fragmentation.

Nesting guilds also differed among treatments. Ground nesters were more common in grassland habitat than the other 3 treatments and were more abundant in the shrub/pole treatment than in fragmented and intact forest. This result was expected because all of our grassland bird species were ground nesters with the exception of the Red-winged Blackbird and the Willow Flycatcher. Shrub nesters were more abundant in the shrub/pole treatment than the other 3 treatments, and were more abundant in grassland than fragmented or intact forest (Table 21). Subcanopy- and cavity-nesting species were more abundant in the fragmented and intact forest treatments than in the shrub/pole or grassland treatments and were more abundant in shrub/pole than grasslands. Canopy-nesting species showed a treatment-by-year interaction. In 1999 they did not differ in abundance between fragmented and intact forest, but in 2000 they were more abundant in intact forest than in fragmented forest (Table 21).

Total abundance and richness also differed among treatments. Abundance and richness were higher in the shrub/pole treatment than any of the other 3 treatments (Table 21). This was expected due to the heterogeneity of the habitat in this treatment which included grass/forbs, shrubs, and small trees. Abundance in fragmented forests did not differ between either intact forest or grassland treatments, but intact forest had higher abundance than grassland habitat (Table 21). Richness did not differ between fragmented and intact forest, but richness in grassland habitat was lower than both of these habitats (Table 21). Similarly, Allaire (1979) found songbird density and richness higher in forested habitat than in grassland habitat in eastern Kentucky, and Willson (1974) found forests and old fields to have higher bird species diversity than grasslands.

Generally, our results comparing habitat guilds among treatments are not unexpected and follow patterns reported in the literature. It is well documented that as vegetative structure and composition change through succession that the corresponding bird community also changes (e.g. Wiens and Rotenberry 1981, James and Wamer 1982).

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Similarity Among Songbird Communities

Fragmented and intact forests shared the highest number of species, and both the Jaccard and Renkonen indices were highest for this pair of treatments (Table 22). Similarity was lowest between grassland and intact forest, and intermediate for the other treatment pairs (Table 22). The grassland/shrub pair also was relatively similar, sharing 12-23 species and having a Jaccard similarity index between 0.40 and 0.48. The grassland areas that we surveyed were not pure stands of grass but also had scattered shrubs and blocks of autumn olive that attracted shrub species. Similarly, the shrub/pole areas we surveyed were adjacent to grassland habitat and often had open patches of grass that were used by grassland birds interspersed among trees. Both fragmented and intact forests shared species with the shrub community. These species were often interior-edge species that use both forest interior as well as edge habitat. Some edge species also were encountered in forested habitats along logging roads, trails, and other gaps in the canopy.

Nesting Success of Grassland Birds

We monitored a total of 36 nests on reclaimed MTMVF areas in 1999-2000 (Table 23), for a total of 308.5 observation days (days that nests were active). Approximately 300 ha of grassland habitat were searched for nests. In 1999 only the Hobet mine was searched for nests, whereas in 2000 all 3 mines were searched.

Overall nest survival of all species combined was 31.1% for the 2 years of the study. Nesting survival in 1999 was only 4.1%, but was higher in 2000 at 52.7%. This difference may be due to the extreme drought conditions in 1999 (Fig. 17). Nest survival in 2000 varied among mine sites, ranging from a low of 1.8% at the Cannelton mine to 68.1% at the Hobet mine (Table 23). Grassland birds had lower nest survival (20.3%) than shrub-nesting birds (48.8%). Shrub nests were found incidentally by nest searchers while searching for grassland bird nests or by other researchers on the project.

More Grasshopper Sparrow nests (19) were found than for any other species (Table 23). Nest survival for this species (36.4%), was similar to that reported in Missouri and Illinois (Table 24), but was higher than other studies. Although density of Grasshopper Sparrow nests was low (~0.06 nests/ha), it was similar to densities on airport grasslands in Illinois and reclaimed mines in northern West Virginia (Table 24). Tallgrass pairie in Oklahoma had much higher nest densities, possibly because this area has the highest abundance of Grasshopper Sparrows and is the center of the species' breeding range (Wells and Rosenberg 1999).

In general, nest densities were low on our study sites. Approximately 537 person-hours were spent nest searching in 2000 by 2 full-time individuals and 3 part-time individuals, and only 25 active nests were located. We do not believe that low nest numbers were a result of nest searchers missing nests. Nest searchers were trained in proper nest searching techniques prior to the start of the study. They searched for nests using standard techniques, including rope dragging, systematically traversing the area and flushing females, and observing parental behavior. Further, the number of nests of Grasshopper Sparrows, our most abundant species in 2000, was similar to the number found by other researchers in other regions of the country in 1 year (Table 24; Wray 1982, Kershner and Bollinger 1996, Koford 1999, McCoy et al. 1999, Rohrbaugh et al. 1999). It is unlikey that nest searchers would miss finding nests of other species if they were able to locate nests of Grasshopper Sparrows, a species known for its ability to conceal its nest (Ehrlich et al. 1988). Habitat measurements surrounding Grasshopper Sparrow nests indicated a high amount of concealment cover around nest sites (Table 25).

Fledgling surveys conducted in late July and early August also indicated that nest densities were low on the mines. Approximately 1.9, 1.7, and 0.4, fledglings/ha were observed in grassland habitat on the Daltex, Hobet, and Cannelton mines, respectively.

There are several possible explanations for low nest densities. First, the habitat may be supporting a biased sex ratio favoring males. Although densities of male Grasshopper Sparrows were high on the mines, few females were observed, suggesting that populations present on these mines included a high proportion of unmated males. Dickcissels are known to have a biased sex ratio favoring males (Buckelew and Hall 1994). Male grasshopper sparrows may have only recently colonized the Daltex mine while females may not have arrived yet. Second, densities of other grassland species, especially Eastern Meadowlarks and Horned Larks, appeared to be relatively low. Our point count abundances included all birds seen or heard, and Eastern Meadowlarks, Horned Larks, and Red-winged Blackbirds were often observed in groups, thus our densities may not represent the number of potential breeding pairs. Also, Red-winged Blackbirds were primarily observed breeding in cattails around ponds and not in the grassland habitat. Since we were primarily concerned with grassland birds, these wetland areas were not as thoroughly searched as the grassland habitat. Lastly, large sections of the mines have been planted with sericea lespedeza which grows in thick, dense stands. A sub-sample of grassland sampling points (n=28) had an average of 21.6% lespedeza cover within the 50-m radius circle, and some sampling points, especially those at the Cannelton mine, had 90-100% lespedeza cover. No grassland bird nests were found in areas with such high lespedeza cover. Grassland birds need areas of open ground with sparse vegetation for foraging and courtship (Whitmore 1979), and areas with thick lespedeza do not appear to provide this requirement. Further, lespedeza cover surrounding Grasshopper Sparrow nests averaged only 4.3% (Table 25), indicating that this species prefers to nest in areas with little lespedeza cover.

Habitat characteristics surrounding Grasshopper Sparrow nests were similar to those reported by Strait (1981). He found grass, shrub, and forb covers surrounding his nests of 32.5, 1.3, and 31.7%, respectively, which are similar to our values of 44.3, 1.7, and 36.3%. Also, the mean vegetation height surrounding his nests was 5.6 dm, which fell within our range of 4.4-5.9 dm. However, he found a deeper litter depth surrounding his nests, at 6.67 cm, whereas ours only ranged from 1.5-2.1 cm (Table 25).

Summary

In summary, MTMVF areas provided breeding habitat for both grassland and early successional species. Grassland, edge, and interior-edge songbirds were more abundant on the post-mining landscape. The highest bird species richness was found in the shrub/pole treatment and the lowest was found in the grassland treatment. Richness in fragmented forest and intact forest fell between these 2 treatments. Ponds on MTMVF areas also provided habitat for waterfowl, wading birds, swallows, and shorebirds, primarily during migration. No federally-listed endangered or threatened species were detected during the study. West Virginia does not have a state threatened and endangered species listing process, but 3 observed grassland species (Grasshopper Sparrow, Henslow's Sparrow, and Bobolink) are considered rare in West Virginia. However, abundances of the forest interior guild and some forest interior species (e.

g. Ovenbird and Acadian Flycatcher) were significantly lower in fragmented forest than in intact forest. Some forest species also were detected more frequently at points further from mine edges. Populations of forest birds will be detrimentally impacted by the loss and fragmentation of mature forest habitat in the mixed mesophytic forest region, which has the highest bird diversity in forested habitats in the eastern United States. Fragmentation-sensitive species such as the Cerulean Warbler, Louisiana Waterthrush, Worm-eating Warbler, Black-and-white Warbler, and Yellow-throated Vireo will likely be negatively impacted as forested habitat is lost and fragmented from MTMVF. Grassland birds nesting on MTMVF areas had nest survival rates similar to those found in the literature, but some species, particularly the Grasshopper Sparrow and Dickcissel, appeared to have high proportions of unmated males in their populations. Further research is necessary to adequately determine the impacts of MTMVF on the nest survival and population dynamics of grassland-nesting bird species.

Raptors

During broadcast surveys, seasonal overall mean abundance for raptors across the 4 treatment types was highest for summer in the grassland treatment (Table 26). Mean abundances separated by mine and treatments are found in Appendix 5. Overall mean abundances for migration in both the grassland and shrub/pole treatments also were greater compared to all other seasons/treatments. Large numbers of Turkey Vultures were observed over grassland and shrub/pole areas during these time periods. Turkey Vultures primarily forage over large open areas, including transitional habitat (Bent 1937, Buckelew and Hall 1994). Overall mean richness was highest in the winter season for the shrub/pole treatment. Five species, including the Northern Harrier, Red-tailed Hawk, Red-shouldered Hawk, Turkey Vulture, and an unidentified *Accipiter*, were detected on surveys in the shrub/pole treatment during winter.

Red-shouldered Hawk abundance was highest in the intact forest treatment during migration and summer. Many studies have shown Red-shouldered Hawks nest primarily in contiguous mature forest habitat (Bednarz and Dinsmore 1981, Morris and Lemon 1983, Belleman 1998). Although most common in intact forest, Red-shouldered Hawks also were recorded in the shrub/pole treatment during all seasons, particularly during migration and winter periods. Some studies have reported greater use of more open areas and woodland edges by Red-shouldered Hawks during the winter months as compared to the summer months (Bohall and Collopy 1984, Crocoll 1994). *Accipiter* species such as Sharp-shinned Hawks also use transitional habitat near open areas during the winter months (Bildstein and Meyer 2000). Northern Harrier and American Kestrel abundances were highest in grasslands, although Northern Harriers also were recorded in the shrub/pole treatment. These 2 species are generally found in more open habitat and rarely are seen over forested habitat except possibly during migration (Johnsgard 1990). Red-tailed Hawks were recorded in every treatment type and were most common in grasslands during the summer months. Several studies have described the Red-tailed Hawk as an open country raptor using agricultural fields, pastures, and forest edges more than other woodland raptor species with little fluctuation in habitat use across seasons (Bent 1937, Bednarz and Dinsmore 1982, Preston and Beane 1993, Moorman and Chapman 1996).

During roadside surveys, overall abundance and richness was highest in the grasslands at the Daltex mine Table 27). Red-tailed Hawks and Turkey Vultures were observed in all 3 treatments during roadside surveys. This is consistent with these species' tendency to forage over expansive open areas and transitional habitats (Bednarz and Dinsmore 1982, Hall 1983). American Kestrels, Northern Harriers, and Broad-winged Hawks were observed in habitats typically frequented by these species. A notable species observed during roadside surveys was a Peregrine Falcon in the grassland at Daltex.

In an overall comparison of raptor species observed on the 3 mines to what would be expected in West Virginia from breeding records and habitat requirements (Table 28), 2 species (Peregrine Falcon and Northern Harrier) unexpectedly occurred on the mines. Two other species, the Rough-legged Hawk and the Short-eared Owl, unexpectedly occurred on the mines during winter.

Even prior to 1950 and the widespread use of DDT, Peregrine Falcons were rare in West Virginia, although there are some nesting records from documented eyries in Mineral, Greenbrier, and Morgan Counties. More recent breeding attempts in the state were recorded in 1991 and 1992 in Grant County after a release of birds in the New River Gorge in 1987-1989 (Buckelew and Hall 1994), and in 2000 with a pair nesting near North Fork Mountain (C. Stihler, personal communication). There are no confirmed breeding records of Peregrine Falcons in Kanawha, Boone, or Logan counties (Buckelew and Hall 1994) and most sightings of Peregrine Falcons in the state have been during migration along mountain ridges (Hall 1983). At least 2 adult Peregrine Falcons were observed throughout the summer months and during the migration season in the grasslands on the Daltex mine. These 2 birds were commonly observed near a rocky "highwall" left after mining activities, but we found no evidence of breeding. An unconfirmed sighting of a Peregrine Falcon occurred during the summer months in the grasslands at the Cannelton mine, but a confirmed sighting of an immature peregrine falcon occurred later during broadcast surveys in November 2000.

Northern Harriers are rare summer/winter residents, but can occasionally be seen in open areas during migration (Hall 1983). There are no breeding records for the species in southwestern West Virginia (Buckelew and Hall 1994). Northern harriers have also been observed in sections of northeastern West Virginia (Canaan Valley) during late summer, migration, and winter (J. Anderson, pers comm.). We observed Northern Harriers in the grasslands during the winter and migration seasons on all 3 mines, and also during the summer months on both the Hobet and Cannelton mines. Northern Harriers also were observed in the shrub treatment at Cannelton during summer and migration. A recent study speculated that reclaimed surface mines may be providing breeding habitat for Northern Harriers, because breeding attempts for Northern Harriers (based on Pennsylvania Breeding Bird Atlas data) were correlated with regions in Pennsylvania containing large numbers of surface mines (Rohrbaugh and Yahner 1996). In other studies, Northern Harriers were commonly observed on surface mines during the breeding season (Yahner and Rohrbaugh 1996, Yahner and Rohrbaugh 1998). Historically, Northern Harriers have occurred in low numbers in West Virginia because of few open areas (wetlands, agricultural lands) for breeding, but recent observations on grassland and shrub/pole areas indicate that Northern Harriers are using reclaimed MTMVF areas in West Virginia, although breeding is not confirmed.

Two winter visitors, the Rough-legged Hawk and the Short-eared Owl also were observed on the mines in open habitats (Table 28). Rough-legged Hawks have been observed in West Virginia during migration along mountain ridges and during winter around Charleston in Kanawha County (Hall 1983). Short-eared Owls are considered rare or uncommon migrants and winter residents in West Virginia due to lack of open habitat such as fields, marshes, and thickets, which this species uses during the nonbreeding season (Hall 1983, Holt and Leasure 1993). Most past sightings of Short-eared Owls occurred in the northern and western counties of West Virginia. Our observation of Short-eared Owls in the grasslands during winter suggests that reclaimed MTMVF areas may be providing wintering habitat for this species.

Broad-winged and Red-shouldered Hawks were observed not only in intact forest as expected in West Virginia, but in forest fragments, shrub/pole areas, and grasslands (Table 28). Broad-winged Hawks and Red-shouldered Hawks are mainly forest species that nest in contiguous mature forest (Crocoll and Parker 1989) although Broad-winged Hawks appear to nest in forests with more openings than Red-shouldered Hawks (Titus and Mosher 1981, Crocoll and Parker 1989). Other studies have shown that Red-shouldered Hawks inhabit more open areas during the winter months (Bohall and Collopy 1984, Peterson and Crocoll 1992). The observations of these 2 species in grassland areas may have been instances where the birds were soaring from 1 forest area to another. In addition, the Red-shouldered Hawk observations could have been territorial displays, because the majority of summer grassland observations occurred during 1999 where the birds were observed soaring extremely high and vocalizing.

Cooper's Hawks and Sharp-shinned Hawks were observed in areas where they were not expected in West Virginia. Cooper's Hawks were sighted in grassland areas during migration. Sharp-shinned Hawks were observed both in grassland during summer and shrub/pole during winter, and an unidentified *Accipiter* species (either Cooper's or Sharp-shinned Hawk) was observed in a forest fragment during winter. There is little habitat information on Cooper's Hawks during migration, but it has been noted that this species uses forest edge as primary hunting habitat in its home range during breeding and uses agricultural fields when overwintering in Texas (Rosenfield and Bielefeldt 1993). Similar to Cooper's Hawks, Sharp-shinned Hawks have been observed in open areas and transitional habitat more during the winter months than summer (Bildstein and Meyer 2000). The observation of a Sharp-shinned Hawk in grasslands during summer may have been a bird passing between forest habitats. It should be noted that most of these unexpected occurrences of a species in a particular habitat were single sightings and thus probably should not be construed as ecologically significant. Finally, the American Kestrel, Red-tailed Hawk, Barred Owl, Eastern Screech Owl, and Turkey Vulture were observed in areas mostly consistent with what was expected in West Virginia.

The Jaccard community similarity index was highest when comparing shrub/pole with fragmented forest (Table 29) and lowest when comparing grassland with either intact forest or fragmented forest treatments. These results are not unexpected based on known habitat requirements of species found in these treatments. With the Renkonen index, the similarity between shrub/pole and fragmented forest dropped considerably and this may be due to the low abundances of the 4 species shared between the 2 treatments. The Renkonen index comparing the shrub/pole and grassland treatments indicated the greatest similarity in species composition of the raptor community.

Summary

MTMVF has had an effect on overall raptor abundance and diversity through a change in the raptor community. Woodland species such as the Red-shouldered Hawk and Broad-winged Hawk were rarely observed in the open grassland and shrub/pole treatments, but more commonly observed in intact forest. Open-country species such as Northern Harriers and American Kestrels were most often observed in grasslands, with no observations occurring in wooded areas. These results suggest that MTMVF is providing a means for an overall shift from a woodland raptor community to a grassland raptor community.

Mammals

Mammal Species Detected

In 1999 and 2000 we captured (through Sherman live trapping or pitfall trapping intended for herpetofaunal species) or observed through incidental sightings 24 of 40 mammal species (excluding bats) thought to occur in our study areas in southern West Virginia (WV GAP analysis, M. Hight pers. comm.) (Table 30). Representatives from 6 orders occurring in southern West Virginia were included in the 24 species recorded.

Six of 10 carnivore species expected to occur on our study area were detected, either by sighting of the animal or by observation of some sign of the animal's presence, such as footprints, scat, or scent (Table 30). Within the grassland treatment, 50% of the carnivore species expected to occur were detected, whereas 44%, 50%, and 20% were detected in the shrub, fragmented forest and intact forest treatments, respectively. Coyote, known to prefer open areas or areas with a diversity of habitats (Whitaker and Hamilton 1998), were detected in every treatment except intact forest. We also had a single sighting of a bobcat on the road beside a fragmented forest. Bobcats use a wide variety of habitats (Lovallo and Anderson 1996), but are secretive and rarely seen, so our sighting should not be viewed as indicative of their habitat use on the mines. Black bears, detected in all 4 treatments, generally have large home ranges spanning multiple habitat types (Landers et al. 1979), which explains our observations of this species. Yearsley and Samuel (1980) found that red fox and gray fox often foraged on reclaimed strip mines in northern West Virginia but were least likely to do so in the summer. The fact that our studies were conducted in the summer and these animals are very secretive may explain why we had only 2 observations of red fox and none of gray fox. Of the other carnivores detected, the raccoon is a habitat generalist that adapts well to human-disturbed landscapes (Burks 1983, Holman 1983), so our encounters with this species in 3 treatments were not surprising. Lastly, we had a single olfactory detection of what was most likely a striped skunk (spotted skunk was not predicted to occur in this area by the WV Gap data) in the shrub/pole treatment. This treatment resembles their preferred habitat of semi-open areas, mixed woods or brush lands (Wade-Smith and Verts 1982).

Four species of carnivores were not observed: the gray fox and 3 members of the weasel family (least weasel, long-tailed weasel, and mink). Each of these species is secretive and primarily nocturnal (King 1989), so one would not necessarily come across them without using methods specifically designed to detect their presence.

Five species of the order Insectivora were expected to occur on our study areas, and all were detected (Table 30). Four shrew species were detected in all 4 treatments: northern short-tailed shrew, masked shrew, smoky shrew, and pygmy shrew. Short-tailed shrew, masked shrew, and pygmy shrew were expected to occur in all treatments as they have broad habitat requirements (George et al. 1986, Kirkland et al. 1987). The smoky shrew, which is reported to select for damp woods (Caldwell and Bryan 1982) was not predicted to occur in grasslands. The fact that summer 2000 was unusually wet (Fig. 17) may have allowed it to use grassland treatments. The only species of mole expected to be present on our study areas, the hairy-tailed mole, was observed on one occasion in fragmented forest. Moles rarely are found above ground, so they are not likely to be captured in traps or observed incidentally.

Ten species of rodent were observed out of 17 expected on our study areas (Table 30). By treatment, we detected 7 species in grassland, 5 in shrub/pole, 7 in fragmented forest, and 5 in intact forest. One of these, the southern bog lemming, was captured in all 4 treatments and is listed as a rare species by the West Virginia Wildlife & Natural Heritage Program (2000). It can exist in a variety of habitats and may be widespread on our study areas due to the virtual absence of the meadow vole, a direct competitor that is believed to displace the bog lemming where they overlap (Krupa and Haskins 1996). Meadow voles did occur in 3 treatments, but at very low numbers.

The Allegheny woodrat was an unexpected capture in shrub/pole areas. The sites were characterized by the presence of a reclaimed drainage ditch filled with large rip-rap boulders shaded by a few trees that lined the channel. This combination of features apparently simulates the natural rock outcrops where woodrats are often found (Balcom and Yahner 1996). It is listed as threatened, endangered, or as a species of special concern in Indiana, Maryland, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Virginia, and West Virginia due to population declines. Prior to the moratorium placed on the endangered species listing process under federal guidelines, this species was designated as a candidate Category II animal in response to apparent population declines in states along the periphery of its range (Balcom and Yahner 1996). When we realized woodrats occurred at some sites, we conducted additional trapping with Tomahawk live traps in another 40 areas of potential habitat, of which 18 were in shrub/pole, 6 were in fragmented forest, 5 were in intact forest, and 11 were around reclaimed-mine ponds. Woodrats were documented at 8 shrub/pole sites, 1 fragmented forest site, and 1 pond, though trapping effort was not equal at each site. In all, 26 woodrats were captured, including 6 adult males, 7 juvenile males, 10 adult females, and 3 juvenile females. Our limited trapping suggests that woodrats have colonized some older reclaimed areas and are breeding there. However, we did not trap extensively for woodrats at rock outcrops in forested habitat so we cannot compare abundances on reclaimed and intact sites.

Several species that were expected to occur in the counties that contained our study areas were not detected by any methods. Four squirrel species, southern flying squirrel, red squirrel, Eastern gray squirrel, and Eastern fox squirrel, were not observed or otherwise detected. The flying squirrel is strictly nocturnal, spending its days in tree cavities or leaf nests (Weigl 1978), habits that make it difficult to observe incidentally. It is possible, however to capture this species in Sherman traps, and it is surprising that none were captured. The red squirrel, gray squirrel, and fox squirrel are diurnal, so they should have been seen or heard if they were common on the mines. Red squirrels are documented in Fayette and Nicholas counties, so they may occur on the Cannelton mine; however, they may not be present on the Hobet and Daltex mines as no records exist of them in Boone and Logan Counties (M. Hight, personal communication). We also did not find southern red-backed voles or golden mice, small rodents that should have been caught in either the Sherman traps or the pitfall traps if they were present on our study sites. Of these, the golden mouse is a more southern species that is not certain to range into the areas where we trapped (M. Hight, personal communication). Southern red-backed voles are associated with mesic high-elevation forests in the Appalachians (Wharton and White 1967). We probably did not trap in their preferred habitat because trapping transects on our study sites were placed near stream channels.

Three additional orders were detected, represented by 4 species. The eastern cottontail, a member of the order Lagomorpha was expected and observed in all 4 treatments, though it was rarely detected in the forest. This is consistent with Chapman et al. (1980), who describe the cottontail as occupying diverse habitats, but not occurring abundantly in deep forests. In the order Artiodactyla, white-tailed deer and wild boar (*Sus scrofa*) were present. Deer were frequently observed in all treatments while wild boar were known to be present based on hunting records as well as a single observation of an animal near a pond. Wild boar are present only in a small portion of southern West Virginia where they were released as a game species by the WV DNR (Igo 1973, Mayer and Brisbin 1991). Lastly, Virginia opossum of the order Didelphimorphia was observed in the 2 forest treatments, though their use of many habitat types (McManus 1974) implies that they probably used the grassland and shrub/pole treatments as well.

Pond Surveys

Ponds, created as part of the reclamation process, were not considered a treatment as they were found within grassland and shrub/pole treatments. Pond surveys were conducted in 2000 to determine if they represented an important landscape feature for wildlife. In 2000, rainfall was plentiful compared to 1999, an extreme drought year (Fig. 17), and so water may not have been limiting to wildlife. The only species detected near ponds that was not detected elsewhere was the wild boar (Table 30), which is associated with watering holes for wallowing (Whitaker and Hamilton 1998). Another animal that was detected during pond surveys was raccoon, a species often found near streams and ponds where they forage for frogs, fish and waterfowl eggs (Llewellyn and Webster 1960). White-tailed deer and their tracks frequently were seen at pond edges; the deer apparently relied on these upland ponds for water while browsing in grasslands, which are located high above streams.

Two species that were expected to occur around ponds that were not detected are muskrat (*Ondatra zibethica*) and beaver (*Castor canadensis*). Many of the mine ponds seem to be ideal muskrat habitat, as they are overgrown with cattails. Muskrat's conical lodges, built of cattails and other wetland vegetation, should have been obvious if they were present, though we did not survey specifically for them. Muskrats also will tunnel into pond banks to den, with tunnel openings discretely located below water level (Whitaker and Hamilton 1998). However, rocky soil around mine ponds makes this an unlikely alternative here. Ponds also seem to provide summer habitat for beaver whose diet during this season consists of aquatic plants, algae, and herbaceous plants (Jenkins 1975). From fall to spring, their diet consists mostly of tree bark (Jenkins 1975). The lack of woody growth around mine ponds and the physical separation of mine ponds from forests by several hundred meters may restrict beaver to wooded areas on the MTMVF landscape.

Small Mammal Trapping

Numerous small mammal species—shrews, voles, and mice—were captured in Sherman live traps or pitfall traps (Table 30). The most common of these were the 2 *Peromyscus* species of mice, the white-footed mouse and the deer mouse. Although the majority (~95%) of *Peromyscus* were thought to be white-footed mice based on field markings, we did not differentiate between the 2 in our analyses because of the difficulty in distinguishing one from the other (Rich et al. 1996). Other small rodents captured included house mouse, woodland jumping mouse, meadow vole, woodland vole, and southern bog lemming. Unexpected captures in Sherman traps were juvenile eastern cottontail rabbits in grassland treatments, juvenile Virginia opossums in fragmented forest and intact forest, and Allegheny woodrats in shrub/pole treatment. Cottontail rabbits and opossums were not expected because of their size relative to trap size while the woodrat was not expected because we did not trap rock outcrops in forests, the habitat with which they are most often associated (Balcom and Yahner 1996). Of the insectivores, only 2 species were caught in Sherman traps: masked shrew and short-tailed shrew. Pitfall trapping accounted for 2 additional species: pygmy shrew and smoky shrew. The majority of shrew captures were by pitfall traps (240 individuals) compared to 40 individuals captured in Sherman traps.

Species Comparisons Among Treatments

Statistical analysis was performed on Sherman trapping results in 3 treatments in 1999 and 4 treatments in 2000. Indices of relative abundance and species richness (Table 31) were compared among the treatments, with each year's data analyzed separately due to the presence of significant ($F = 9.60$, $df = 2$, $P = 0.0001$) year by treatment interactions. Mean abundances separated by mine and treatment are found in Appendix 6. Reclaimed pond indices (Table 31) were not compared statistically to the other treatments for 2 reasons. First, it was not truly a treatment because the ponds were distributed throughout the reclaimed mines, overlapping both shrub/pole and grassland treatments. Second, sampling methods were different from the other treatments.

In 1999, species richness ranged from 1.7 species per transect in the grassland to 2.3 species per transect in the intact forest with no significant difference ($F = 2.61$, $df = 2$, $P = 0.09$) among treatments (Table 31). There were, however, differences in species composition among treatments as indicated by the Jaccard and Renkonen indices of species similarity (Table 32). In 2000, when shrub/pole areas were added as a fourth treatment, species richness ranged from 1.4 species per transect in the grassland, fragmented forest, and intact forest treatments to 1.5 species in the shrub/pole treatment. Again, there were no significant differences ($F = 0.17$, $df = 3$, $P = 0.92$) among treatments. Richness averaged over all treatments was compared between years as well. Richness in 1999 was 1.9 species per transect compared to 1.4 species per transect in 2000, a significant difference ($F = 19.86$, $df = 1$, $P < 0.0001$). This difference may be explained by changes in weather patterns between years (Gentry et al. 1966). From May through August in 1999, an extreme drought year, there was a total of 29.2 cm of rain in Charleston (Fig. 17), which is the nearest NOAA weather station to the mines we sampled. In 2000, however, 47.0 cm of rain were recorded in Charleston during the same months. Average daily high temperatures also were different between years, with 1999 having an average daily high of 29.1 C° from May to August and 2000 averaging 26.9 C° during those same months (Fig. 18). The thirty-year normal for the 4-month period is 40.8 cm of rain and an average daily high of 27.9 C° (Figs. 17 and 18).

The fact that richness indices were not significantly different among treatments in either year does not mean that the small mammal communities were the same. To compare the species composition between treatments, we calculated Jaccard and Renkonen indices of community similarity (Nur et al. 1999) (Table 32). In 1999, the Jaccard indices, which are based on the number of species shared between treatments but do not take into account species abundances, showed that the 2 forest treatments, fragments and intact, were more similar to each other than either was to the grassland treatment. Similar results were found in 2000, although the differences were not as pronounced. Also, the 2000 Jaccard indices showed that shrub/pole was more similar to grassland than it was to either of the 2 forest treatments. The Renkonen indices were in agreement with each of the trends shown by the Jaccard indices. However, this index, which incorporates similarities in species abundance as well as species composition between treatments, showed a high degree of similarity between treatments being compared. This is probably because *Peromyscus* species accounted for the vast majority of captures in all treatments.

Total relative abundance ($F = 1.42$, $df = 2$, $P = 0.25$) and *Peromyscus* species abundance ($F = 1.79$, $df = 2$, $P = 0.18$) did not differ among the 3 treatments sampled in 1999 (Table 31). In 2000, significant differences were found among treatments for both total abundance ($F = 23.34$, $df = 3$, $P < 0.001$) and *Peromyscus* species abundance ($F = 21.57$, $df = 3$, $P < 0.001$). In each case the grassland and shrub/pole treatments were similar, but had significantly greater abundances than fragmented forest and intact forest, which were similar to each other (Table 32). Because *Peromyscus* represent the majority of the captures, trends in its abundance are the driving factor in the difference found in overall abundance. Other studies on strip mines have shown that *Peromyscus* abundance is highest in early stages of succession (Verts 1957, Sly 1976, Hansen and Warnock 1978). Similarly, *Peromyscus* abundance has been shown to be higher in forest openings created by clearcutting than in adjacent forested areas in the southern Appalachians (Kirkland 1977, Buckner and Shure 1985).

In each year of the study, differences were found among treatments for several individual species captured. House mouse, for example, was captured only in the grassland treatment in both years, a finding consistent with other studies. In addition to human dwellings and other buildings, the house mouse has been found in grassy fields and croplands but almost never in forests (Kaufman and Kaufman 1990, Whitaker and Hamilton 1998). The woodland jumping mouse was captured only in fragmented forest and intact forest. As its name suggests, this species is generally a forest dweller, and is often found near streams (Whitaker and Hamilton 1998). It was found more frequently in fragmented forest than in intact forest. It has been reported to use habitat at the interface between forest and clearing, even venturing into open glades (Whitaker and Wrigley 1972), but no data could be found confirming that it selects for forest edge over interior forest. Except for a single grassland capture, eastern chipmunk also was found primarily in the 2 forest treatments, with intact having a greater abundance than fragmented ($F = 11.20$, $df = 2$, $P < 0.0001$). This result was not necessarily expected, as chipmunks are known to frequent forest edge habitats (Pyare et al. 1993). In 1999, short-tailed shrews differed in abundance between treatments ($F = 4.59$, $df = 2$, $P = 0.016$) with higher abundance in intact forest than in grasslands. Throughout its range, this species uses a variety of habitats, but is known to be restricted to moist woods in Indiana, Kentucky, and Tennessee (Whitaker and Hamilton 1998).

We also found several between-year differences in small mammal abundance. Total abundance in grassland habitats increased from 1999 to 2000 ($F = 4.98$, $df = 1$, $P = 0.03$). The difference may be related to weather patterns, as the combination of drought and high temperatures in summer 1999 may have made it a difficult season to exist in the open grasslands. Lewellen and Vessey (1998) reported that population growth in white-footed mice was negatively correlated with extreme weather conditions in both summer and winter. Fragmented forest ($F = 14.71$, $df = 1$, $P < 0.0001$) and intact forest ($F = 34.40$, $df = 1$, $P < 0.0001$) had decreases in total abundance from 1999 to 2000. This may have been due to the dry, hot weather of 1999 that forced small mammals into the woods in search of water and relief from the high temperatures (Fig. 18), or alternatively, the cool, wet conditions in 2000 made the forest a more extreme environment than the reclaimed areas.

Other species differed in abundance between the 2 years. The number of short-tailed shrew captures dropped from 35 in 1999 to 2 in 2000. Decreased reproduction during the summer 1999 drought may be the cause of this trend. Short-tailed shrews, having a high rate of evaporation from the skin (George et al. 1986), are known to be unable to tolerate hot and dry conditions. Other studies also have noted wide yearly fluctuations in the abundance of this species, but the reason for this is not well understood (Lindeborg 1941, Fowle and Edwards

1955). Woodland jumping mice were caught at the rate of 0.5 individuals per 100 trap nights in intact forests in 2000 after not being caught at all in that treatment in 1999. However, this may not represent an actual difference because each of the individuals caught in intact forest in 2000 was trapped at a single site, one that was not trapped in 1999. Captures of woodland jumping mice also increased slightly in fragmented forest from 1999 to 2000. Southern bog lemmings were trapped in 2000 but not 1999. There is no clear reason for this, though only 2 were trapped in 2000 so the difference most likely does not represent an actual abundance difference between the years.

We also compared the results of our study with those of other small mammal studies conducted in grassland and shrub/pole habitat types (Table 33). However, interpretations of these comparisons should be made with caution for several reasons. First, capture methods differ among the studies, with the majority using snap traps rather than live traps. Capture methods have been shown to affect trapping success (Goodnight and Koestner 1942, Cockrum 1947, and Sealander and James 1958). Second, none of these studies was performed on a reclaimed MTMVF area. Most were on reclaimed strip mines, which may undergo a similar pattern of succession starting with reclamation, but differ from MTMVF areas in that the disturbance occurs on a much smaller spatial scale. A third reason that comparisons with other studies can be misleading is that abundance estimates may be calculated differently. Nelson and Clark (1973) recommended the use of a correction for sprung traps when calculating abundances. We employed this correction, but other studies, especially those prior to 1973, did not correct. In order to make comparisons with these studies, we also have listed our abundances calculated without the correction (Table 33).

Some additional differences between our results and those of other studies can be attributed to geographic differences, as the composition of small mammal communities varies by region. For example, in two of the studies to which we compared our results, those by Clark et al. (1998) in Oklahoma and Sietman et al. (1994) in Kansas, the cotton rat (*Sigmodon hispidus*) was the most abundant small mammal. The fact that they found *Peromyscus* at a much lower abundance than we did may simply be the result of competition with the cotton rat, a species that does not occur on our study areas. Also, the abundance of meadow voles in our grasslands was considerably lower than many of the other studies. For example, it was the most abundant small mammal captured by Mindell (1978) and Forren (1981) in northern West Virginia. It may not be as common in the southern part of the state due to the predominance of forest.

Summary

Our study is in agreement with most literature surveyed in that we found small mammals to be more abundant at early stages of succession than in forest. This trend in our study was driven by the white-footed mouse, a species that is often most abundant in early successional stages (e.g. Hansen and Warnock 1978, Buckner and Shure 1985). Two species, short-tailed shrew and eastern chipmunk, were more abundant in intact forest than fragmented forest. Allegheny woodrats were captured at several shrub/pole sites where rock drains with large boulders and some canopy cover provided useable habitat.

Herpetofauna

Based on habitat requirements and known records of herpetofaunal species reported in Green and Pauley (1987) and personnel communication with T. Pauley, we estimated that 59 species could be expected to occur on our study areas (Table 34), including 39 species that are predominantly terrestrial and 20 species that are predominantly aquatic. Through captures in drift fence arrays, occasional stream searches near arrays, and incidental observations, 35 (59%) species were found on our study areas, most in traps associated with drift fence arrays. No species federally-listed as endangered or threatened or state-listed as species of concern were found. Terrestrial and aquatic species of salamanders were least represented. Of the 39 terrestrial species expected to occur, we found 24 species (62%). We found 33% of species expected to occur within the grassland treatment, 81% within the shrub/pole treatment, 47% within forest fragments, and 53% within intact forests. Less developed vegetative cover and thick homogenous plantings of lespedeza likely resulted in the low value for the grassland treatment.

Only data from drift fence arrays were subjected to statistical analyses. Mean richness ($F=1.40$, $df=3$, $P=0.25$) and abundance ($F=1.14$, $df=3$, $P=0.34$) of all herpetofaunal captures combined did not differ between the 4 treatments (Table 35). We found no interactions between treatment and sampling period (richness: $F=0.69$, $df=15$, $P=0.78$; abundance: $F=0.61$, $df=15$, $P=0.85$). The number of different species captured ranged from 13 in young reclaimed grassland treatment to 16 in the fragmented forest treatment. In a study comparing herpetofaunal populations in recent clearcuts and mature forests, Pais et al. (1988) found that overall abundance did not differ between their treatments. Their study was conducted in eastern Kentucky where the herpetofaunal community is similar to our study sites and they used similar sampling methods (drift fence arrays). Thus, response of herpetofauna in overall abundance was similar in disturbed and undisturbed sites, whether the disturbance resulted from timber harvesting or from mining. However, Pais et al. (1988) found lowest species richness in their mature forest treatment, while we found no differences between treatments. As expected on our study sites, the herpetofaunal community was most similar between the grassland and shrub/pole treatments and most dissimilar between the grassland and intact forest treatments (Table 36).

Salamanders comprised about a quarter of individuals and species captured in fragmented and intact forest (Table 37). They were less common in the grassland and shrub/pole treatments, both in number of species and individuals. Red-spotted newts, both the adult and juvenile (red eft) forms were the most common species and the most widely distributed (Table 38). Both adults and juveniles were captured in all 4 treatments and at every sampling point. The only salamander species captured outside of the 2 forested treatments was a spotted salamander in a grassland array. Green and Pauley (1987) indicate that this species is typically found in deciduous forests but has been documented in newly plowed fields. In a review of 18 studies of amphibian responses to clearcutting, a disturbance that results in early successional habitats, de Maynadier and Hunter (1995) found that amphibian abundance was 3.5 times higher in unharvested stands than in recent clearcuts. So it was not surprising to find few salamanders in our early successional habitats. In 2-yr-old clearcuts in eastern Kentucky (an area with a herpetofaunal community similar to southern West Virginia), Pais et al. (1988), captured 5 species of salamanders with drift fence arrays. Their clearcuts (12-15 ha) were much smaller than our reclaimed sites and had forested habitat in closer proximity, which probably contributed to differences in salamander richness. Additionally, greater amounts of woody debris ground cover, higher soil moisure, and looser soil likely contributed to higher salamander richness in their early successional habitats (clearcuts) compared to ours (reclaimed mines). DeMaynadier and Hunter (1998) found that lack of canopy cover, litter cover, and cover from snags, stumps, and associated root channels potentially limited amphibians near forest edges created by clearcutting.

Toads and frogs were captured in high numbers in all 4 treatments, ranging from 53% to 72% of all individuals captured within a treatment (Table 37). High numbers of these species were captured during the August and September trapping periods and included many individuals that had recently metamorphosed, particularly green and pickerel frogs (Table 38). Summer of 2000 was an abnormally wet year (Fig. 17) and standing water occurred throughout the treatments providing ample habitat for breeding. The eastern American toad, green frog, and pickerel frog occurred at almost every sample point and within each treatment (Table 38). The wood frog, which typically occurs in moist, deciduous forests (Green and Pauley 1987), was captured only in the intact forest treatment.

Three species of lizards were captured in arrays; all were captured in low numbers and at few sample points (Table 38). Although only 5 species of lizards occur in southern West Virginia (Green and Pauley 1987), we had expected to capture them in greater numbers. The fence lizard in particular is known to occur in xeric habitats and was captured only in grassland and shrub/pole treatments. Because this species typically does not occur in moist forest conditions, it probably was not abundant on the study sites before mining occurred. It is not known how long it would take this species to colonize reclaimed mine sites since surrounding lands are generally forested. The ground skink, categorized by West Virginia Natural Heritage Program (2000) as a rare ("S3") species, was found only in the intact forest treatment. This species generally inhabits the floor of dry, open woodlands and uses leaf litter and decaying wood for concealment and foraging (Conant 1975, Green and Pauley 1987)

Only 1 species of turtle, the box turtle, was captured in the arrays and it occurred in all treatments except shrub/pole (Table 38). This was the only species of terrestrial turtle expected to occur within our study areas. Turtle species generally are not sampled well by drift fence arrays, so captures of box turtles probably are not representative of the actual population.

Snakes were the most common group captured in grassland and shrub/pole habitats, ranging from 46-50% of species captured within these 2 treatments (Table 37). Within fragmented forest and intact forest, snakes accounted for 26-31% of species captured. Snakes are very mobile and may be able to colonize reclaimed sites more quickly than other herpetofaunal species and generally tolerate drier habitats resulting in the higher proportion of snake species. The total number of species and individuals was higher in the shrub/pole sites than in the forested sites. Similarly, Ross et al. (2000) found fewer species of snakes in forested areas with high tree densities. Two species were captured exclusively in the forest treatments, worm snake, and redbelly snake (Table 38). The worm snake is considered a rare ("S3") species by the West Virginia Natural Heritage Program (2000). Green and Pauley (1987) state that redbelly snakes frequent open forests and forest edges and the species appears to prefer mountainous terrain. Similarly, eastern worm snakes prefer forest lands. This species frequently burrows in decayed logs or underground, so it is not surprising that this species was not captured in the reclaimed grassland or shrub/pole treatments. Three species, hognose (also classed as a rare "S3" species), black racer, and northern water snake, were captured only in the 2 reclaimed treatments. The hognose and black racer are known to frequent dry, open sites. The northern water snake will occur in almost any habitat if there is a reasonable amount of water (Green and Pauley 1987), and the wet summer during 2000 provided such areas in the reclaimed grassland and shrub/pole treatments.

Summary

The herpetofaunal community sampled from March through September 2000, shifted from a majority of amphibian species in the 2 forested treatments to a majority of reptile species in the grassland and shrub/pole treatments. In particular, salamander species decreased while snake species increased. Summer 2000 had much more rainfall than normal (see mammal results section) which provided ample breeding habitat for toads and frogs, a group that accounted for a high proportion of species and individuals in all treatments. Thus, we may have found a more pronounced shift during a drier summer. Herpetofaunal species that require loose soil, moist conditions, and woody or leaf litter ground cover generally were absent from reclaimed sites. Minimizing soil compaction, establishing a diverse vegetative cover, and adding coarse woody debris to reclaimed sites would provide habitat for some herpetofaunal species more quickly after mining. Salamander populations, however, appear to require several years to recover in areas disturbed by clearcutting (50-70 years: Petranka et al. 1993; 20-24 years: Ash 1997). MTMVF results in greater soil disturbance than clearcutting so a longer time may be required for recovery of salamander populations in reclaimed mine sites.

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Table 1. Watersheds and stream drainages with songbird (S), raptor (R), mammal (M), and herpetofaunal (H) sampling points by treatment in 3 watersheds in southwest West Virginia.

		Treatment			
Watershed	Streams	Grassland	Shrub/pole	Fragmented Forest	Intact Forest
Mud River	Big Horse	SRM		SRMH	
	Lavender Fork	SRMH		SMH	
	Stanley Fork	SRM		SRM	
	Spring Branch				SRMH
	Big Buck Fork				SR
	Hill Fork		SRM		
	Long Branch		SRMH		
Spruce Fork	Rockhouse Creek	SRMH			
	Bend Branch				SRM
	Beech Creek			SRM	
	Pigeonroost Branch				SRH
Twentymile Creek	Bullpush Fork	SRMH	SRMH		
	Ash Fork				SRMH
	Hughes Fork			SRMH	

Table 2. Number of replicates in each treatment and watershed for each taxa in 2000.

		Treatment			
Taxa	Watershed	Grassland	Shrub/pole	Fragmented Forest	Intact Forest
Songbirds	Mud River	18	17	20	20
	Spruce Fork	12	0	6	17
	Twentymile Creek	10	16	10	10
Mammals	Mud River	6	4	6	6
	Spruce Fork	2	0	2	2
	Twentymile Creek	2	4	2	2
Raptors	Mud River	4	6	4	5
	Spruce Fork	4	0	4	4
	Twentymile Creek	4	6	4	3
Herps	Mud River	1	1	2	1
	Spruce Fork	1	0	0	1
	Twentymile Creek	1	2	1	1

Table 3. Mean and range of estimated age and elevation of grassland, shrub/pole, fragmented forest, and intact forest treatments and total area of each treatment at each mine site.

	Treatment							
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest	
Mine	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Age (yrs)								
Hobet 21	12	8-14	16	16	--[a]	--	--	--
Daltex	8	5-11	--	--	--	--	--	--
Cannelton	13	9-19	23	13-27	--	--	--	--
Elevation (m)								
Hobet 21	367	304-423	322	241-375	308	253-358	328	276-406
Daltex	424	341-516	--	--	343	299-452	440	358-533
Cannelton	444	388-476	439	382-467	374	332-428	477	360-566
Area (ha)	Total	Range	Total	Range	Total	Range	Total	Range
Hobet 21	2003	--	428	--	339	83-157	--	--
Daltex	1819	--	106[b]	--	155	30-86	--	--
Cannelton	1672	--	508	--	214	--	--	--

[a] Data not applicable to this treatment or mine site.
[b] This shrub/pole habitat was not used for the study because it did not result from MTMVF.

Table 4. Codes for wind speed, sky cover, and edge types used in point count surveys.

Wind Speed	Sky Cover	Edge Types
0 = Smoke rises vertically	0 = Clear or few clouds	1 = Paved road
1 = Wind direction shown by smoke	1 = Partly cloudy	2 = Open-canopy road
2 = Wind felt on face, leaves rustle	2 = Cloudy or overcast	3 = Partially open-canopy road
3 = Leaves, small twigs in constant motion	3 = Fog	4 = Agricultural opening
4 = Raises dust and loose paper, small branches move	4 = Drizzle	5 = Development (houses, etc.)
5 = Small trees in leaf sway	5 = Showers	6 = River or stream
		7 = Clearcut
		8 = Wildlife opening
		9 = Natural gap
		10 = Valley Fill
		11 = Grassland
		12 = Forest
		13 = Pond
		14 = Autumn Olive Block

Table 5. Partner-in Flight (PIF) conservation ratings and action levels for upland forest birds in the Ohio Hills physiographic area, the percent of each species' population estimated to be within that area, the percent of forested point counts where these species were detected during this study, and species for which logistic regression models were developed.

Species	PIF rating[a]	Action level[ab]	Percent of population[ac]	Percent of point counts[d]	Logistic Regression Model?
Cerulean Warbler	30	II	46.8	36.1	yes
Swainson's Warbler	25	IV	1.9	1.2	no
Louisiana Waterthrush	25	III	11.6	15.7	yes
Worm-eating Warbler	24	IV	12.5	21.7	yes
Kentucky Warbler	22	IV	11.2	26.5	yes
Acadian Flycatcher	22	IV	15.6	81.9	yes
Eastern Wood-pewee	21	III	3.4	1.2	no
Wood Thrush	21	IV	9.1	56.6	yes
Yellow-throated Vireo	21	IV	8.5	20.5	yes
Hooded Warbler	21	IV	8.0	38.5	yes
Black-billed Cuckoo	21	IV	1.9	0.00	no
Scarlet Tanager	19	IV	11.1	47.0	yes
Great Crested Flycatcher	19	IV	1.0	1.2	no
Yellow-billed Cuckoo	19	IV	<1.0	9.6	no
Black-and-white Warbler	19	IV	1.3	41.0	yes

[a] Draft PIF Landbird Conservation Plan: Physiographic Area 22: Ohio Hills (Rosenburg 2000).
[b] Action levels: I=crisis; recovery needed; II=immediate management or policy needed rangewide; III=management to reverse or stabilize populations; IV= long-term planning to ensure stable populations; V=research needed to better define threats; VI=monitor population changes only (Rosenburg 2000).
[c] Percent of population thought to occur in the Ohio Hills area 22 calculated from percent of range area, weighted by BBS relative abundance (Rosenberg 2000).
[d] Percent of forested point counts (n=83) where species occurred in 1999-2000.

Table 6. Mean and standard error (SE) for habitat variables measured at grassland (n=44), shrub/pole (n=33), fragmented forest (n=36), and intact forest (n=49) sampling points.

	Treatment							
	Grassland		Shrub/Pole		Fragmented Forest		Intact Forest	
Variables	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Slope (%)	16.96	2.10	10.16	1.93	33.78	2.28	33.75	2.07
Aspect Code	1.05	0.10	0.78	0.13	1.05	0.12	1.02	0.08
Grass/Forb Height (dm)	7.29	0.27	6.20	0.48	--[a]	--	--	--
Litter Depth (cm)	2.26	0.19	1.64	0.17	--	--	--	--
Elevation (m)	400.93	7.19	378.85	11.53	332.08	7.11	389.58	10.87
Distance to Minor Edge (m)	113.02	16.75	68.14	8.23	38.71	3.88	64.61	11.57
Distance to Habitat Edge (m)	335.46	45.26	79.16	11.06	128.61	12.52	1430.66	145.32
Distance to Forest/Mine Edge (m)	347.35	44.30	253.98	34.46	128.61	12.52	1430.66	145.32
Robel Pole Index	2.93	0.17	4.30	0.27	--	--	--	--
Canopy Height (m)	--	--	4.67	0.45	21.70	0.72	22.90	0.67
<u>Ground Cover (%)</u>								
Water	0.14	0.10	0.15	0.12	1.15	0.32	0.48	0.17
Bareground	7.73	1.18	2.22	0.92	7.71	0.95	7.45	0.59
Litter	8.14	1.54	6.06	1.78	54.24	1.88	48.32	1.75
Woody Debris	0.06	0.04	0.30	0.12	4.20	0.42	4.95	0.41
Moss	1.04	0.38	1.83	0.86	2.01	0.32	2.04	0.34
Green	82.77	2.00	85.86	3.47	30.35	1.74	36.61	1.99
Forb Cover	23.63	2.39	21.89	2.86	--	--	--	--
Grass Cover	45.05	2.71	43.70	5.26	--	--	--	--
Shrub Cover	14.13	2.72	22.99	3.23	--	--	--	--
<u>Stem Densities (no./ha)</u>								
<2.5 cm	777.70	207.52	2590.91	351.50	2034.72	119.64	1670.92	100.40
>2.5-6 cm	73.15	18.79	993.37	151.95	6439.24	537.40	7122.45	741.86
>8-23 cm	0.85	0.43	113.26	20.71	374.65	37.20	304.08	14.32
>23-38 cm	0.00	0.00	27.65	6.29	93.23	5.60	94.13	5.11
>38-53 cm	0.00	0.00	3.98	1.65	32.29	3.32	31.89	2.60
>53-68 cm	0.00	0.00	1.70	0.87	11.28	1.69	7.91	1.22
>68 cm	0.00	0.00	0.00	0.00	4.34	0.93	3.57	0.73
<u>Canopy Cover (%)</u>								
>0.5-3 m	--	--	29.70	2.94	54.90	2.33	47.63	2.33
>3-6 m	--	--	22.88	2.86	66.63	2.42	54.67	2.06
>6-12 m	--	--	14.37	2.59	63.06	2.38	65.46	1.24
>12-18 m	--	--	2.84	0.86	56.01	2.68	63.34	2.07
>18-24 m	--	--	0.11	0.08	41.39	2.97	51.28	3.06
>24 m	--	--	0.00	0.00	16.15	2.48	18.06	2.14
Structural Diversity Index	--	--	3.85	0.29	11.58	0.23	11.37	0.22

[a] Variables were not measured in this treatment.

Table 7. Two-way ANOVA results comparing habitat variables among treatments and mines.

	Factor Levels															
	Treatment			Waller-Duncan[a]				Mine			Waller-Duncan[b]			Treatment x Mine		
Variables	F	df	P	GR	SH	FR	IN	F	df	P	Can.	Dal.	Hob.	F	df	P
Slope (%)	39.79	3	<0.01	B	C	A	A	26.55	2	<0.01	B	A	A	5.26	5	<0.01
Aspect Code	2.07	3	0.11					0.05	2	0.95				1.90	5	0.10
Elevation (m)	24.94	3	<0.01	A	B	C	A	106.18	2	<0.01	A	B	C	4.63	5	<0.01
Grass Height (dm)	3.82	1	0.06					20.78	2	<0.01	C	B	A	4.26	1	0.04
Litter Depth (cm)	3.56	1	0.06					25.07	2	<0.01	C	B	A	2.31	1	0.13
Distance to minor edge (m)	4.69	3	<0.01	A	B	B	B	0.35	2	0.70				2.08	5	0.07
Distance to habitat edge (m)	647.34	3	<0.01	B	C	C	A	184.31	2	<0.01	B	A	C	185.51	5	<0.01
Distance to mine/forest edge (m)	537.85	3	<0.01	B	C	D	A	142.67	2	<0.01	B	A	C	172.57	5	<0.01
Robel Pole Index	20.66	1	<0.01					11.09	2	<0.01	A	B	C	0.00	1	0.94
Canopy Height (m)	222.33	2	<0.01	--	B	A	A	1.02	2	0.36				7.66	3	<0.01
Ground Cover (%):																
Water	5.87	3	<0.01	C	C	A	B	1.26	2	0.28				0.40	5	0.85
Bareground	14.55	3	<0.01	A	B	A	A	3.91	2	0.02	AB	A	B	2.30	5	0.05
Litter	208.5	3	<0.01	C	C	A	B	4.14	2	0.02	C	A	B	9.24	5	<0.01
Woody Debris	121.45	3	<0.01	B	B	A	A	2.41	2	0.09				0.95	5	0.45
Moss	4.61	3	<0.01	B	B	A	A	0.24	2	0.79				0.95	5	0.45
Green	119.75	3	<0.01	B	A	C	C	2.18	2	0.12				1.63	5	0.15
Forb	0.07	1	0.79					4.99	2	0.01	A	A	B	3.56	1	0.06
Grass	0.15	1	0.70					22.22	2	<0.01	B	B	A	4.93	1	0.03
Shrub	3.54	1	0.06					14.68	2	<0.01	A	B	B	4.52	1	0.04
Stem Density (no./ha):																
<2.5 cm	51.56	3	<0.01	B	A	A	A	4.39	2	0.01				5.80	5	<0.01
>2.5-8 cm	196.94	3	<0.01	C	B	A	A	2.90	2	0.06				2.07	5	0.07
>8-23 cm	514.48	3	<0.01	C	B	A	A	3.28	2	0.04				1.09	5	0.37
>23-38 cm	276.56	3	<0.01	C	B	A	A	0.00	2	0.99				0.31	5	0.91
>38-53 cm	189.33	3	<0.01	C	B	A	A	0.71	2	0.49				3.26	5	<0.01
>53-68 cm	31.73	3	<0.01	C	C	A	B	0.87	2	0.42				1.88	5	0.10
>68 cm	13.35	3	<0.01	B	B	A	A	2.25	2	0.11				1.56	5	0.17

Table 7. Continued.

	Factor Levels															
	Treatment			Waller-Duncan[a]				Mine			Waller-Duncan[b]			Treatment x Mine		
Variables	F	df	P	GR	SH	FR	IN	F	df	P	Can.	Dal.	Hob.	F	df	P
Canopy Cover (%):																
0.5-3 m	24.15	2	<0.01	--	C	A	B	0.98	2	0.38				1.69	3	0.17
>3-6 m	69.44	2	<0.01	--	C	A	B	0.10	2	0.91				3.68	3	0.01
>6-12 m	144.61	2	<0.01	--	B	A	A	0.02	2	0.98				1.85	3	0.14
>12-18 m	259.89	2	<0.01	--	C	B	A	0.82	2	0.44				0.65	3	0.58
>18-24 m	154.75	2	<0.01	--	C	B	A	1.95	2	0.15				1.82	3	0.15
>24 m	30.83	2	<0.01	--	B	A	A	1.41	2	0.25				2.58	3	0.06
Structural Diversity Index	262.81	2	<0.01	--	B	A	A	0.09	2	0.91				2.38	3	0.07

[a] Waller-Duncan k-ratio t-test. Treatments with different letters differ at P<0.05 ('A' indicates highest value). GR=grassland; SH=shrub/pole; FR=fragmented forest; IN=intact forest.
[b] Waller-Duncan k-ratio t-test. Mines with different letters differ at P<0.05 ('A' indicates highest value). Can.=Cannelton; Dal.=Daltex; Hob.=Hobet.

Table 8. ANOVA results comparing habitat variables among mines within individual treatments for variables with treatment x mine interactions.

	Treatment/Mine																						
	Grassland			Waller-Duncan[a]			Shrub/pole			Waller-Duncan		Fragmented Forest			Waller-Duncan			Intact Forest			Waller-Duncan		
Variables	F	df	P	Can.	Dal.	Hob.	F	df	P	Can.	Hob.	F	df	P	Can.	Dal.	Hob.	F	df	P	Can.	Dal.	Hob.
Slope (%)	2.30	2	0.11	B	A	AB	120.21	1	<0.01	B	A	6.40	2	<0.01	B	A	A	4.72	2	0.01	B	B	A
Aspect Code	1.84	2	0.17				2.93	1	0.09	B	A	0.47	2	0.63				1.03	2	0.36			
Elevation (m)	19.53	2	<0.01	A	A	B	127.50	1	<0.01			14.40	2	<0.01	A	B	C	37.36	2	<0.01	A	B	C
Distance to habitat edge (m)	15.69	2	<0.01	B	A	B	3.40	1	0.07	A	B	3.60	2	0.04	AB	B	A	445.12	2	<0.01	A	A	B
Distance to forest/mine edge (m)	13.72	2	<0.01	B	A	B	11.33	1	<0.01	B	A	3.60	2	0.04	AB	B	A	445.12	2	<0.01	A	A	B
Grass Height (dm)	5.42	2	<0.01	B	B	A	31.76	1	<0.01	B	A	--	--	--				--	--	--			
Canopy Height (m)	--	--	--				1.21	1	0.28			7.29	2	<0.01	A	B	B	3.17	2	0.05	AB	A	B
Ground Cover (%):																							
Bareground	3.75	2	0.03	AB	A	B	0.77	1	0.39			4.00	2	0.03	A	B	B	0.59	2	0.56			
Litter	12.35	2	<0.01	C	B	A	6.24	1	0.02	A	B	1.92	2	0.16				5.72	2	<0.01	B	A	B
Grass	9.73	2	<0.01	B	B	A	25.30	1	<0.01	B	A	--	--	--									
Shrub	13.11	2	<0.01	AB	B	C	5.95	1	0.02	A	B	--	--	--									
Stem Density (no./ha):																							
<2.5cm	5.81	2	<0.01	B	A	A	0.00	1	0.98			2.07	2	0.14				0.07	2	0.93			
>38-53cm	--	--	--				3.47	1	0.07	A	B	1.36	2	0.27				5.16	2	<0.01	B	A	A
Canopy Cover (%):																							
>3-6m	--	--	--				2.63	1	0.11			0.28	1	0.76				6.00	2	<0.01	A	A	B
	--	--	--																				
Structural Diversity Index	--	--	--				1.38	1	0.25			0.33	1	0.72				3.30	2	0.05	AB	A	B

[a] Waller-Duncan k-ratio t-test. Mines with different letters differ at P<0.05 (‘A’ indicates highest value). Can.=Cannelton; Dal.= Daltex; Hob.=Hobet.

Table 9. ANOVA results comparing habitat variables among treatments at individual mines for variables with treatment x mine interactions.

	Mine/treatment																			
	Cannelton			Waller-Duncan[a]				Daltex			Waller-Duncan			Hobet			Waller-Duncan			
Variables	F	df	P	GR	SH	FR	IN	F	df	P	GR	FR	IN	F	df	P	GR	SH	FR	IN
Slope (%)	39.47	3	<0.01	B	C	A	A	1.77	2	0.19				22.80	3	<0.01	B	B	A	A
Aspect Code	4.06	3	0.01	A	B	A	AB	1.00	2	0.38				0.10	3	0.96				
Elevation (m)	11.28	3	<0.01	AB	B	C	A	9.18	2	<0.01	A	B	A	11.93	3	<0.01	A	BC	C	B
Distance to habitat edge (m)	759.76	3	<0.01	B	B	B	A	209.89	2	<0.01	B	C	A	18.43	3	<0.01	B	C	B	A
Distance to forest/mine edge (m)	660.78	3	<0.01	B	B	B	A	209.89	2	<0.01	B	C	A	8.04	3	<0.01	BC	BA	C	A
Grass Height (dm)	4.25	1	0.05					--	--	--				0.01	1	0.91				
Canopy Height (m)	97.45	1	<0.01	--	B	A	A	--	--	--				123.98	2	<0.01	--	B	A	A
Ground Cover (%):																				
Bareground	7.33	3	<0.01	A	B	A	A	1.58	2	0.22				8.94	3	<0.01	B	C	AB	A
Litter	50.67	3	<0.01	C	B	A	A	173.58	2	<0.01	B	A	A	101.76	3	<0.01	C	D	A	B
Grass	3.70	1	0.07					--	--	--				1.64	1	0.21				
Shrub	0.03	1	0.86					--	--	--				12.34	1	<0.01				
Stem Densities (no./ha):																				
<2.5cm	50.28	3	<0.01	B	A	A	A	13.42	2	<0.01	B	A	A	8.48	3	<0.01	B	A	A	A
>38-53cm	39.03	3	<0.01	D	C	A	B	91.33	2	<0.01	B	A	A	134.64	3	<0.01	B	B	A	A
Canopy Cover (%):																				
>3-6m	29.42	2	<0.01	--	B	A	A							35.47	2	<0.01	--	C	B	A
Structural Diversity Index	117.12	2	<0.01	--	B	A	A							194.46	2	<0.01	--	C	A	B

[a] Waller-Duncan k-ratio t-test. Treatments with different letters differ at P<0.05 (‘A’ indicates highest value). GR=grassland; SH=shrub/pole; FR=fragmented forest; IN=intact forest.

Table 10. Mean distance from subplot centers to minor edge types within treatments, and the percentage of subplots within each treatment that were closest to that edge type.

	Grassland			Shrub/pole			Fragmented Forest			Intact Forest		
	Distance (m)			Distance (m)			Distance (m)			Distance (m)		
Minor Edge Type	Mean	SE	Percent	Mean	SE	Percent	Mean	SE	Percent	Mean	SE	Percent
Paved road	40.00	0.00	0.63	--	--	0.00	--	--	0.00	--	--	0.00
Open-canopy road	105.97	14.71	40.51	76.10	6.02	73.23	54.03	4.56	24.65	57.10	7.37	10.26
Partially-open canopy road	--	--	0.00	--	--	0.00	12.72	3.78	12.68	58.96	7.12	48.21
Stream	--	--	0.00	--	--	0.00	35.99	3.80	47.89	34.77	4.01	31.79
Natural gap/wildlife opening	--	--	0.00	--	--	0.00	34.00	7.97	3.52	11.50	8.50	1.03
Valley fill	118.80	16.97	55.70	36.36	6.46	19.69	38.40	13.38	3.52	--	--	0.00
Grassland	--	--	0.00	--	--	0.00	77.50	2.50	1.41	--	--	0.00
Forest	44.00	16.99	3.16	75.71	13.38	5.51	--	--	0.00	--	--	0.00
Pond	--	--	0.00	10.00	5.00	1.57	--	--	0.00	--	--	0.00
Combination	--	--	0.00	--	--	0.00	35.00	7.45	6.34	239.71	28.91	8.72

Table 11. Comparison of species found to be “probable” or “confirmed” breeders in southwestern West Virginia by the West Virginia Breeding Bird Atlas (WV BBA) or expected to be there by the West Virginia Gap Analysis Lab (Gap), and those observed during this study during surveys and/or incidentally (x=observed during breeding season, m=assumed to be migrating).

			This Study				
Species	WV BBA	Gap	Grassland	Shrub/ pole	Fragmented Forest	Intact Forest	Pond
Forest Interior Species							
Acadian Flycatcher	x	x		x	x	x	
Black-throated Blue Warbler		x					
Black-throated Green Warbler	x	x				x	x
Blue-headed Vireo	x	x			x	x	
Canada Warbler		x					
Cerulean Warbler	x	x		x	x	x	
Eastern Wood-pewee	x	x				x	
Great Crested Flycatcher	x	x				x	
Kentucky Warbler	x	x			x	x	
Louisiana Waterthrush	x	x			x	x	
Ovenbird	x	x		x	x	x	
Pileated Woodpecker	x	x			x	x	
Scarlet Tanager	x	x		x	x	x	x
Summer Tanager	x	x			x	x	
Swainson’s Warbler	x	x			x		
Veery		x				x	
Winter Wren	x	x					
Wood Thrush	x	x			x	x	
Worm-eating Warbler	x	x			x	x	
Yellow-throated Warbler	x	x			x	x	
Interior-edge Species							
American Redstart	x	x			x	x	
American Robin	x	x			x	x	x
Black-and-white-Warbler	x	x			x	x	
Black-billed Cuckoo	x	x	x	x	x		
Black-capped Chickadee	x	x			x	x	
Blue-gray Gnatcatcher	x	x			x	x	
Carolina Chickadee	x	x	x	x	x	x	x
Carolina Wren	x	x		x	x	x	
Common Raven	x	x	x				x
Dark-eyed Junco		x			x		
Downy Woodpecker	x	x		x	x	x	x
Eastern Phoebe	x	x	x	x		x	x
Eastern Towhee	x	x	x	x		x	x
Hairy Woodpecker	x	x		x	x	x	
Hooded Warbler	x	x		x	x	x	
Least Flycatcher		x					
Northern Flicker	x	x	x	x	x	x	
Northern Parula	x	x		x	x	x	
Palm Warbler							m

Table 11. Continued.

Species	WV BBA	Gap	This Study				
			Grassland	Shrub/ pole	Fragmented Forest	Intact Forest	Pond
Pine Warbler	x	x					
Red-bellied Woodpecker	x	x			x	x	
Red-eyed Vireo	x	x	x	x	x	x	x
Red-headed Woodpecker		x					
Rose-breasted Grosbeak		x					
Ruby-throated Hummingbird	x	x	x	x	x	x	
Ruffed Grouse	x	x			x	x	
Tufted Titmouse	x	x		x	x	x	
Whip-poor-will	x	x				x	
White-breasted Nuthatch	x	x		x	x	x	
Wild Turkey	x	x	x	x	x	x	x
Yellow-billed Cuckoo	x	x	x	x	x	x	x
Yellow-throated Vireo	x	x			x	x	x
Edge Species							
American Crow	x	x			x		
American Goldfinch	x	x	x	x	**x**	x	x
American Woodcock	x	x			x	x	x
Baltimore Oriole	x	x	x		x		x
Blue Grosbeak		x	x	x	x		x
Blue Jay	x	x	x		x	x	x
Blue-winged Warbler	x	x	x	x	x	x	x
Brown Thrasher	x	x	x	x			x
Brown-headed Cowbird	x	x				x	
Cedar Waxwing	x	x	x	x	x		
Chestnut-sided Warbler		x					
Chipping Sparrow	x	x		x			
Common Grackle	x	x					
Common Yellowthroat	x	x	x	x			x
Eastern Bluebird	x	x	x	x			x
Eastern Kingbird	x	x	x				
Field Sparrow	x	x	x	x			x
Golden-winged Warbler	x	x		x			x
Gray Catbird	x	x		x			x
House Wren	x	x					
Indigo Bunting	x	x	x	x	x	x	x
Mourning Dove	x	x	x	x			x
Northern Bobwhite	x	x	x				
Northern Cardinal	x	x	x	x	x	x	x
Northern Mockingbird	x	x		x			
Orchard Oriole	x	x	x	x			x
Prairie Warbler	x	x	x	x			x
Purple Finch				x			
Song Sparrow	x	x	x	x	x		x
Warbling Vireo	x	x					
White-eyed Vireo	x	x		x	x		
Yellow Warbler	x	x	x	x			x

Table 11. Continued.

Species	WV BBA	Gap	This Study Grassland	Shrub/ pole	Fragmented Forest	Intact Forest	Pond
Yellow-breasted Chat	x	x	x	x	x		x
Grassland Species							
Bobolink			x				m
Dickcissel			x				x
Eastern Meadowlark	x	x	x	x			x
Grasshopper Sparrow		x	x	x			x
Henslow's Sparrow		x	x				x
Horned Lark		x	x				x
Red-winged Blackbird	x	x	x	x	x		x
Ring-necked Pheasant			x				
Vesper Sparrow			x				
Willow Flycatcher		x	x				x
Wetland Species							
American Black Duck		x					
American Bittern							x
Blue-winged Teal		x					m
Canada Goose	x	x	x		x		x
Common Merganser							m
Double-crested Cormorant						m	
Great Blue Heron							x
Green Heron	x	x					x
Hooded Merganser		x					
Mallard	x	x	x				x
Spotted Sandpiper		x					m
Swamp Sparrow		x					
Wood Duck	x	x			x		x
Greater Yellowlegs							**m**
Lesser Yellowlegs							m
Least Sandpiper							m
Pied-billed Grebe							m
Solitary Sandpiper							m
White-rumped Sandpiper							m
Green-winged Teal							m
Yellow-crowned Night-heron						m	
Other Species							
Bank Swallow		x					
Barn Swallow	x	x	x		x		x
Belted Kingfisher	x	x			x	x	
Chimney Swift	x	x	x		x		x
Cliff Swallow		x	x				
Common Nighthawk	x	x	m				
European Starling	x		x				
House Finch	x	x					

Table 11. Continued.

			This Study				
Species	WV BBA	Gap	Grassland	Shrub/ pole	Fragmented Forest	Intact Forest	Pond
House Sparrow	x						
Killdeer	x	x	x				x
Northern Rough-winged Swallow	x	x	x				x
Purple Martin	x	x					
Tree Swallow	x	x	x		x		x
Rock Dove	x						

Table 12. Bird species observed (means with standard errors in parentheses) during 50-m radius point count surveys on reclaimed MTMVF areas in grassland, shrub/pole, fragmented forests, and intact forest treatments in Boone, Fayette, Kanawha, and Logan Counties, West Virginia, 1999-2000.

	Treatment								ANOVA Results[a]	
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest			
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Forest Interior Species										
Acadian Flycatcher	0.00	0.00	0.17	0.03	0.96	0.86	1.11	1.32	4.87	0.03
	(0.00)	(0.00)	(0.17)	(0.03)	(0.15)	(0.11)	(0.12)	(0.12)		
Black-throated Green	0.00	0.00	0.00	0.00	0.00	0.17	0.06	0.17	0.21	0.65
Warbler	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.06)	(0.04)	(0.06)		
Blue-headed Vireo	0.00	0.00	0.00	0.00	0.25	0.19	0.44	0.36	2.86	0.09
	(0.00)	(0.00)	(0.00)	(0.00)	(0.09)	(0.08)	(0.12)	(0.08)		
Cerulean Warbler	0.00	0.00	0.00	0.03	0.21	0.31	0.36	0.36	1.22	0.27
	(0.00)	(0.00)	(0.00)	(0.03)	(0.08)	(0.10)	(0.11)	(0.09)		
Eastern Wood-pewee	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.02		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.03)	(0.02)		
Great Crested Flycatcher	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.02)		
Kentucky Warbler	0.00	0.00	0.17	0.00	0.29	0.25	0.28	0.26	0.00	0.97
	(0.00)	(0.00)	(0.17)	(0.00)	(0.11)	(0.08)	(0.09)	(0.08)		
Louisiana Waterthrush	0.00	0.00	0.00	0.00	0.08	0.19	0.17	0.06	1999:0.58	1999: 0.45
	(0.00)	(0.00)	(0.00)	(0.00)	(0.06)	(0.07)	(0.07)	(0.04)	2000:3.33	2000: 0.07
Ovenbird	0.00	0.00	0.00	0.03	0.54	0.61	1.00	1.34	18.03	<0.01
	(0.00)	(0.00)	(0.00)	(0.03)	(0.10)	(0.10)	(0.13)	(0.17)		
Pileated Woodpecker	0.00	0.03	0.00	0.00	0.17	0.08	0.00	0.06	1999:6.96	1999: 0.01
	(0.00)	(0.03)	(0.00)	(0.00)	(0.08)	(0.05)	(0.00)	(0.04)	2000:0.11	2000: 0.74
Scarlet Tanager	0.00	0.00	0.00	0.09	0.21	0.31	0.11	0.68	1999:1.22	1999: 0.27
	(0.00)	(0.00)	(0.00)	(0.05)	(0.08)	(0.10)	(0.07)	(0.12)	2000:6.03	2000: 0.02
Summer Tanager	0.00	0.00	0.00	0.00	0.13	0.08	0.11	0.13	0.08	0.78
	(0.00)	(0.00)	(0.00)	(0.00)	(0.07)	(0.05)	(0.05)	(0.05)		
Swainson's Warbler	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.03)	(0.00)	(0.00)		
Wood Thrush	0.00	0.00	0.00	0.00	0.79	0.36	0.44	0.64	0.08	0.77
	(0.00)	(0.00)	(0.00)	(0.00)	(0.18)	(0.09)	(0.11)	(0.12)		
Worm-eating Warbler	0.00	0.00	0.00	0.00	0.08	0.19	0.19	0.17	0.25	0.62
	(0.00)	(0.00)	(0.00)	(0.00)	(0.06)	(0.07)	(0.08)	(0.06)		

Table 12. Continued.

	Treatment								ANOVA Results[a]	
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest			
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Yellow-throated Warbler	0.00	0.00	0.00	0.00	0.04	0.17	0.08	0.09	0.14	0.71
	(0.00)	(0.00)	(0.00)	(0.00)	(0.04)	(0.07)	(0.06)	(0.04)		
Interior-edge Species										
American Redstart	0.00	0.00	0.50	0.06	0.25	0.25	0.53	0.77	13.21	<0.01
	(0.00)	(0.00)	(0.22)	(0.04)	(0.11)	(0.07)	(0.09)	(0.13)		
American Robin	0.00	0.00	0.00	0.03	0.04	0.00	0.00	0.02		
	(0.00)	(0.00)	(0.00)	(0.03)	(0.04)	(0.00)	(0.00)	(0.02)		
Black-and-white Warbler	0.00	0.00	0.00	0.03	0.29	0.28	0.22	0.34	0.00	0.98
	(0.00)	(0.00)	(0.00)	(0.03)	(0.09)	(0.09)	(0.07)	(0.07)		
Black-capped Chickadee	0.00	0.00	0.00	0.00	0.04	0.00	0.03	0.02		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.04)	(0.00)	(0.03)	(0.02)		
Blue-gray Gnatcatcher	0.00	0.00	0.00	0.00	0.04	0.00	0.03	0.11		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.04)	(0.00)	(0.03)	(0.09)		
Carolina Chickadee	0.00	0.03	0.00	0.27	0.42	0.42	0.42	0.28	0.57	0.45
	(0.00)	(0.03)	(0.00)	(0.10)	(0.12)	(0.12)	(0.12)	(0.08)		
Carolina Wren	0.00	0.00	0.17	0.03	0.38	0.19	0.44	0.06	0.23	0.63
	(0.00)	(0.00)	(0.17)	(0.03)	(0.12)	(0.07)	(0.11)	(0.04)		
Dark-eyed Junco	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.04)	(0.00)	(0.00)	(0.00)		
Downy Woodpecker	0.00	0.00	0.00	0.18	0.08	0.28	0.06	0.00	1999: 0.17	1999: 0.68
	(0.00)	(0.00)	(0.00)	(0.08)	(0.06)	(0.09)	(0.04)	(0.00)	2000:12.33	2000:<0.01
Eastern Phoebe	0.00	0.00	0.00	0.15	0.00	0.00	0.00	0.04		
	(0.00)	(0.00)	(0.00)	(0.06)	(0.00)	(0.00)	(0.00)	(0.03)		
Eastern Towhee	0.03	0.08	0.50	0.76	0.00	0.00	0.00	0.02		
	(0.03)	(0.04)	(0.34)	(0.11)	(0.00)	(0.00)	(0.00)	(0.02)		
Hairy Woodpecker	0.00	0.00	0.00	0.09	0.00	0.06	0.11	0.09	2.11	0.15
	(0.00)	(0.00)	(0.00)	(0.05)	(0.00)	(0.04)	(0.05)	(0.05)		
Hooded Warbler	0.00	0.00	0.33	0.03	0.17	0.14	0.42	0.57	13.07	<0.01
	(0.00)	(0.00)	(0.21)	(0.03)	(0.08)	(0.07)	(0.10)	(0.10)		
Northern Flicker	0.00	0.03	0.00	0.06	0.08	0.00	0.06	0.02		
	(0.00)	(0.03)	(0.00)	(0.04)	(0.06)	(0.00)	(0.06)	(0.02)		
Northern Parula	0.00	0.00	0.00	0.03	0.17	0.36	0.14	0.11	1999:0.01	1999: 0.92
	(0.00)	(0.00)	(0.00)	(0.03)	(0.10)	(0.09)	(0.06)	(0.05)	2000:7.19	2000: <0.01
Red-bellied Woodpecker	0.00	0.00	0.00	0.00	0.04	0.08	0.08	0.09	1999:0.39	1999: 0.53
	(0.00)	(0.00)	(0.00)	(0.00)	(0.04)	(0.05)	(0.05)	(0.04)	2000:0.00	2000: 0.98

Table 12. Continued.

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		ANOVA Results[a]	
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Red-eyed Vireo	0.00	0.03	0.50	0.42	1.00	1.72	0.92	1.38	3.30	0.07
	(0.00)	(0.03)	(0.22)	(0.10)	(0.12)	(0.14)	(0.13)	(0.11)		
Ruby-throated	0.00	0.03	0.00	0.06	0.08	0.11	0.11	0.04		
Hummingbird[b]	(0.00)	(0.03)	(0.00)	(0.04)	(0.06)	(0.07)	(0.05)	(0.03)		
Tufted Titmouse	0.00	0.00	0.00	0.09	0.13	0.28	0.17	0.23	0.00	0.99
	(0.00)	(0.00)	(0.00)	(0.05)	(0.07)	(0.08)	(0.06)	(0.06)		
White-breasted Nuthatch	0.00	0.00	0.00	0.03	0.08	0.19	0.22	0.15	0.39	0.53
	(0.00)	(0.00)	(0.00)	(0.03)	(0.08)	(0.07)	(0.08)	(0.05)		
Yellow-billed Cuckoo	0.00	0.03	0.33	0.06	0.04	0.14	0.08	0.00	1999:0.39	1999: 0.53
	(0.00)	(0.03)	(0.21)	(0.04)	(0.04)	(0.06)	(0.05)	(0.00)	2000:7.40	2000: <0.01
Yellow-throated Vireo	0.00	0.00	0.00	0.00	0.13	0.22	0.08	0.11	1.81	0.71
	(0.00)	(0.00)	(0.00)	(0.00)	(0.07)	(0.07)	(0.05)	(0.05)		
Edge Species										
American Crow[b]	0.00	0.00	0.00	0.09	0.13	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.00)	(0.05)	(0.09)	(0.00)	(0.00)	(0.00)		
American Goldfinch	0.37	0.25	2.67	0.55	0.08	0.14	0.00	0.02	1999:3.16	1999:0.08
	(0.14)	(0.07)	(1.73)	(0.14)	(0.06)	(0.09)	(0.00)	(0.02)	2000:2.04	2000:0.16
Baltimore Oriole	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Blue Grosbeak	0.00	0.15	0.00	0.06	0.00	0.00	0.00	0.00		
	(0.00)	(0.07)	(0.00)	(0.04)	(0.00)	(0.00)	(0.00)	(0.00)		
Blue Jay[b]	0.03	0.00	0.00	0.00	0.08	0.08	0.03	0.11		
	(0.03)	(0.00)	(0.00)	(0.00)	(0.06)	(0.06)	(0.03)	(0.05)		
Blue-winged Warbler	0.10	0.00	1.17	0.48	0.04	0.00	0.06	0.00		
	(0.06)	(0.00)	(0.17)	(0.11)	(0.04)	(0.00)	(0.04)	(0.00)		
Brown Thrasher	0.10	0.08	0.17	0.06	0.00	0.00	0.00	0.00		
	(0.07)	(0.04)	(0.17)	(0.04)	(0.00)	(0.00)	(0.00)	(0.00)		
Brown-headed Cowbird	0.00	0.00	0.00	0.00	0.00	0.03	0.06	0.15	3.42	0.07
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.03)	(0.06)	(0.05)		
Cedar Waxwing[b]	0.00	0.13	0.00	0.33	0.00	0.00	0.00	0.00		
	(0.00)	(0.09)	(0.00)	(0.13)	(0.00)	(0.00)	(0.00)	(0.00)		
Chipping Sparrow	0.00	0.00	0.17	0.27	0.00	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.17)	(0.08)	(0.00)	(0.00)	(0.00)	(0.00)		

Table 12. Continued.

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		ANOVA Results[a]	
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Common Yellowthroat	0.37	0.15	0.50	0.79	0.00	0.00	0.00	0.00		
	(0.10)	(0.07)	(0.34)	(0.12)	(0.00)	(0.00)	(0.00)	(0.00)		
Eastern Bluebird	0.00	0.03	0.00	0.06	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.04)	(0.00)	(0.00)	(0.00)	(0.00)		
Field Sparrow	0.37	0.68	1.00	1.27	0.00	0.00	0.00	0.00		
	(0.12)	(0.16)	(0.26)	(0.21)	(0.00)	(0.00)	(0.00)	(0.00)		
Golden-winged Warbler	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.00)	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)		
Gray Catbird	0.00	0.00	0.17	0.15	0.00	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.17)	(0.06)	(0.00)	(0.00)	(0.00)	(0.00)		
Indigo Bunting	0.80	0.98	0.83	1.70	0.17	0.19	0.03	0.06	6.68	0.01
	(0.16)	(0.13)	(0.31)	(0.19)	(0.08)	(0.07)	(0.03)	(0.04)		
Mourning Dove	0.07	0.08	0.00	0.09	0.00	0.00	0.00	0.00		
	(0.07)	(0.04)	(0.00)	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)		
Northern Bobwhite[b]	0.03	0.08	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.03)	(0.04)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Northern Cardinal	0.00	0.03	0.50	0.24	0.08	0.17	0.00	0.04		
	(0.00)	(0.03)	(0.22)	(0.08)	(0.06)	(0.08)	(0.00)	(0.04)		
Orchard Oriole	0.00	0.05	0.00	0.18	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.09)	(0.00)	(0.00)	(0.00)	(0.00)		
Prairie Warbler	0.10	0.23	0.67	1.15	0.00	0.00	0.00	0.00		
	(0.06)	(0.08)	(0.21)	(0.15)	(0.00)	(0.00)	(0.00)	(0.00)		
Song Sparrow	0.20	0.23	0.00	0.09	0.04	0.03	0.00	0.00		
	(0.10)	(0.09)	(0.00)	(0.05)	(0.04)	(0.03)	(0.00)	(0.00)		
White-eyed vireo	0.07	0.08	0.33	0.45	0.00	0.03	0.00	0.00		
	(0.05)	(0.04)	(0.21)	(0.10)	(0.00)	(0.03)	(0.00)	(0.00)		
Yellow Warbler	0.30	0.08	0.33	0.27	0.00	0.03	0.00	0.00		
	(0.09)	(0.04)	(0.21)	(0.11)	(0.00)	(0.03)	(0.00)	(0.00)		
Yellow-breasted Chat	0.23	0.15	0.67	1.33	0.00	0.06	0.00	0.00		
	(0.08)	(0.06)	(0.21)	(0.16)	(0.00)	(0.04)	(0.00)	(0.00)		
Grassland Species										
Bobolink	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		

Table 12. Continued.

	Treatment								ANOVA Results[a]	
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest			
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Dickcissel	0.20	0.18	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.12)	(0.08)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Eastern Meadowlark	0.63	0.58	0.00	0.06	0.00	0.00	0.00	0.00		
	(0.17)	(0.13)	(0.00)	(0.04)	(0.00)	(0.00)	(0.00)	(0.00)		
Grasshopper Sparrow	2.23	2.95	0.33	0.27	0.00	0.00	0.00	0.00		
	(0.19)	(0.22)	(0.33)	(0.09)	(0.00)	(0.00)	(0.00)	(0.00)		
Henslow's Sparrow	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Horned Lark	0.33	0.23	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.09)	(0.08)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Red-winged Blackbird	1.37	0.73	0.00	0.36	0.00	0.03	0.00	0.00		
	(0.28)	(0.21)	(0.00)	(0.16)	(0.00)	(0.03)	(0.00)	(0.00)		
Vesper Sparrow	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Willow Flycatcher	0.13	0.15	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.06)	(0.06)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Other Species										
American Kestrel[b]	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Barn Swallow[b]	0.00	0.05	0.00	0.03	0.00	0.00	0.00	0.00		
	(0.00)	(0.03)	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)		
Belted Kingfisher[b]	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.03)	(0.00)		
Chimney Swift[b]	0.00	0.18	0.00	0.30	0.00	0.00	0.00	0.00		
	(0.00)	(0.15)	(0.00)	(0.12)	(0.00)	(0.00)	(0.00)	(0.00)		
Cliff Swallow[b]	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Cooper's Hawk[b]	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00		
	(0.00)	(0.00)	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)		
European Starling[b]	0.00	0.40	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.40)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Killdeer[b]	0.13	0.08	0.17	0.00	0.00	0.00	0.00	0.00		
	(0.06)	(0.04)	(0.17)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		

Table 12. Continued.

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		ANOVA Results[a]	
Species	1999	2000	1999	2000	1999	2000	1999	2000	F	P
Mallard[b]	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.07)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Northern Rough-winged	0.00	0.48	0.00	0.03	0.00	0.00	0.00	0.00		
Swallow[b]	(0.00)	(0.15)	(0.00)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)		
Tree Swallow[b]	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.00)	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Turkey Vulture[b]	0.03	0.05	0.00	0.00	0.00	0.00	0.03	0.00		
	(0.03)	(0.03)	(0.00)	(0.00)	(0.00)	(0.00)	(0.03)	(0.00)		
Unknown Bird[b]	0.07	0.10	0.17	0.24	0.21	0.11	0.11	0.15		
	(0.05)	(0.05)	(0.17)	(0.08)	(0.08)	(0.05)	(0.05)	(0.05)		
Unknown Sparrow[b]	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00		
	(0.05)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Unknown Swallow[b]	0.50	0.00	0.33	0.00	0.00	0.00	0.00	0.00		
	(0.26)	(0.00)	(0.33)	(0.00)	(0.00)	(0.00)	(0.00)	(0.00)		
Unknown Woodpecker[b]	0.00	0.00	0.00	0.00	0.08	0.11	0.06	0.02		
	(0.00)	(0.00)	(0.00)	(0.00)	(0.06)	(0.05)	(0.04)	(0.02)		

[a] ANOVA results testing for differences in species abundances between fragmented and intact forest. Only species observed at >5% of point counts were analyzed.

[b] Not used in subsequent analyses of songbird richness, similarity, or total abundance.

Table 13. Comparison of bird densities (birds/ha) in grassland habitats of the United States.

	Study									
	This study	Allaire (1979)	Wray (1982)	DeVault et al. (in review)	Warren & Anderson (unpub. data)	Vickery et al. (1999)	Wiens (1973)	Wiens (1973)	Norment et al. (1999)	Frawley and Best (1991)
Location	SW West Virg.	E. Kent.	N.West Virg.	SW Ind.	NE West Virg. (high elev.)	Maine	Western U.S.	Western U.S.	W. New York	Iowa
Survey Method[a]	PC	ST/SM	TF	RPC	ST	SM	TF	TF	PC	SM
Habitat[b]	MTM	MTM	SM	SM	PA/WM	GR	GR-grazed	GR-ungrazed	GR/PA	AF – unmowed
Species	Density[c]									
Bobolink	0.00-0.03	0.00-0.00	nr	0.00-0.01	0.42	0.00-0.15	nr	nr	0.00-6.37	nr
Dickcissel	0.12-0.25	0.00-0.00	nr	0.09-0.34	0.00	0.00-0.70	0.81	nr	0.00-0.00	0.01
Eastern Meadowlark	0.50-0.70	0.07-0.38	nr	0.39-0.79	0.13	0.00-0.15	0.88	nr	0.00-0.64	nr
Grasshopper Sparrow	2.49-2.80	0.17-0.40	1.23-1.53	0.25-0.51	0.02	0.00-0.35	0.38-0.74	0.19-1.54	0.00-0.00	0.01
Horned Lark	0.21-0.33	0.02-0.19	0.23-0.55	0.04-0.05	0.00	0.00-0.25	0.18-1.97	0.49-1.20	0.00-0.01	nr
Red-winged Blackbird	0.83-1.17	0.12-0.33	nr	0.67-1.29	0.19	nr	nr	nr	0.00-0.02	0.40
Savannah Sparrow	0.00-0.00	0.00-0.00	0.65-1.10	0.00-0.01	0.22	0.00-0.35	nr	nr	0.00-3.82	nr
Vesper Sparrow	0.00-0.00	0.00-0.00	0.87-0.97	0.00-0.00	0.00	0.20-0.45	0.54	nr	0.00-0.00	0.10
Total Abundance	10.27-10.54	0.35-1.06	nr	nr	nr	nr	nr	nr	0.00-10.19	nr
Richness	1-12	2-5	nr	nr	nr	nr	4-6	3-10	0-4	8

[a] PC=point count; ST=strip transect; SM=spot mapping; RPC=roadside point count; TF=territory flush. Note: territory flush and spot mapping are meaures of territory density, not bird density.
[b] MTM=mountaintop mining/valley fill, SM=surface mine, GR=natural grassland, PA=pasture, WM=wet meadow; AF=alfalfa field.
[c] Range represents minimum and maximum values reported; single values indicate an average value; nr=not reported.

Table 14. Species abundance, total abundance, richness, and similarity in the shrub/pole treatment in areas that were relatively young (13-25 years old; n=27) and in areas that were older (>26 year old; n=6) in 2000 compared to Denmon's (1998) study in early successional habitats of West Virginia.

	Treatment				
	Young Shrub/pole		Old Shrub/pole		
Species	Mean	SE	Mean	SE	Denmon (1998)
Acadian Flycatcher	0.04	0.04	0.00	0.00	0.09
American Goldfinch	0.41	0.14	1.17	0.31	0.29
American Redstart	0.07	0.05	0.00	0.00	0.24
American Robin	0.00	0.00	0.17	0.17	0.34
Black-and-white Warbler	0.04	0.04	0.00	0.00	0.10
Blue Grosbeak	0.07	0.05	0.00	0.00	0.00
Blue-winged Warbler	0.44	0.11	0.67	0.33	0.24
Brown Thrasher	0.07	0.05	0.00	0.00	0.03
Carolina Chickadee	0.15	0.07	0.83	0.40	0.11
Carolina Wren	0.04	0.04	0.00	0.00	0.06
Cerulean Warbler	0.04	0.04	0.00	0.00	0.00
Chipping Sparrow	0.30	0.09	0.17	0.17	0.24
Common Yellowthroat	0.89	0.13	0.33	0.21	0.50
Downy Woodpecker	0.22	0.10	0.00	0.00	0.07
Eastern Bluebird	0.07	0.05	0.00	0.00	0.03
Eastern Meadowlark	0.07	0.05	0.00	0.00	0.00
Eastern Phoebe	0.11	0.06	0.33	0.21	0.00
Eastern Towhee	0.63	0.11	1.33	0.21	0.91
Field Sparrow	1.37	0.24	0.83	0.31	0.66
Golden-winged Warbler	0.11	0.06	0.00	0.00	0.09
Grasshopper Sparrow	0.30	0.10	0.17	0.17	0.03
Gray Catbird	0.19	0.08	0.00	0.00	0.33
Hairy Woodpecker	0.11	0.06	0.00	0.00	0.01
Hooded Warbler	0.04	0.04	0.00	0.00	0.06
Indigo Bunting	1.78	0.19	1.33	0.56	1.07
Mourning Dove	0.07	0.05	0.17	0.17	0.00
Northern Cardinal	0.19	0.08	0.50	0.22	0.31
Northern Flicker	0.07	0.05	0.00	0.00	0.01
Northern Parula	0.04	0.04	0.00	0.00	0.00
Orchard Oriole	0.22	0.11	0.00	0.00	0.03
Ovenbird	0.04	0.04	0.00	0.00	0.23
Prairie Warbler	1.15	0.16	1.17	0.48	0.33
Red-eyed Vireo	0.41	0.11	0.50	0.22	1.39
Red-winged Blackbird	0.44	0.19	0.00	0.00	0.06

Table 14. Continued.

Species	Treatment: Young Shrub/pole Mean	SE	Treatment: Old Shrub/pole Mean	SE	Denmon (1998)
Scarlet Tanager	0.07	0.05	0.17	0.17	0.06
Song Sparrow	0.11	0.06	0.00	0.00	0.26
Tufted Titmouse	0.11	0.06	0.00	0.00	0.14
White-breasted Nuthatch	0.04	0.04	0.00	0.00	0.01
White-eyed Vireo	0.52	0.11	0.17	0.17	0.46
Yellow Warbler	0.33	0.13	0.00	0.00	0.37
Yellow-billed Cuckoo	0.04	0.04	0.17	0.17	0.01
Yellow-breasted Chat	1.37	0.19	1.17	0.31	0.54
Richness	9.52	0.39	8.67	0.49	9.80
Total Abundance	12.78	0.68	11.33	1.17	13.40
Species Shared[a]	18				
Jaccard Index[a]	0.45				
Renkonen Index[a]	0.65				

[a] Comparing young and old shrub areas only.

Table 15. Comparison of abundances of common songbird species in different areas of intact forest in the mixed mesophytic forest region.

	This Study	Allaire (1979)	Demeo (1999)	Wood et al. (1998)	Anderson & Shugart (1974)
Location	SW West Virg.	E. Kent.	N. Cen.. West Virg.	N. Cen. West Virg.	E. Tenn.
Survey Method[a]	PC	ST	PC	PC	SM[b]
Habitat[c]	MF	MF	MF	MF	PF/MF
Species	Abundance[d]				
Acadian Flycatcher	1.11-1.32	0.80-0.94	0.50	0.77	Rare
American Redstart	0.53-0.77	0.40-0.74	0.54	0.61	nr
Black-and-white Warbler	0.22-0.34	0.20-0.33	0.39	0.09	nr
Black-throated Green Warbler	0.06-0.17	0.60-0.74	0.58	0.91	nr
Blue-headed Vireo	0.36-0.44	nr	0.30	0.29	nr
Carolina Chickadee	0.28-0.42	0.20-0.53	nr	nr	Very common
Carolina Wren	0.06-0.44	0.07-0.20	nr	nr	Rare
Cerulean Warbler	0.36	0.60-0.74	0.13	0.07	Rare
Downy Woodpecker	0.00-0.06	0.13-0.33	nr	0.13	Common
Hairy Woodpecker	0.09-0.11	0.00-0.07	nr	0.05	Rare
Hooded Warbler	0.42-0.57	0.40-0.80	0.28	0.61	Common
Indigo Bunting	0.03-0.06	nr	nr	0.18	Rare
Kentucky Warbler	0.26-0.28	0.27-0.67	nr	0.05	Rare
Louisiana Waterthrush	0.06-0.17	0.00-0.13	0.07	0.00	nr
Northern Parula	0.11-0.14	0.07-0.13	nr	0.02	nr
Ovenbird	1.00-1.34	0.60-0.67	nr	0.29	Rare
Pileated Woodpecker	0.00-0.06	0.00-0.07	nr	0.05	nr
Red-bellied Woodpecker	0.08-0.09	0.07-0.33	nr	nr	Common
Red-eyed Vireo	0.92-1.38	0.87-1.34	1.41	1.70	Very common
Scarlet Tanager	0.11-0.68	0.13-0.27	0.49	0.54	Commmon
Summer Tanager	0.11-0.13	0.13-0.20	nr	nr	Rare
Tufted Titmouse	0.17-0.23	0.33-0.60	nr	0.00	Very common
White-breasted Nuthatch	0.15-0.22	0.00-0.20	nr	0.14	Common
Wood Thrush	0.44-0.64	0.13-0.53	0.38	0.57	Rare
Worm-eating Warbler	0.17-0.19	0.53-0.87	0.07	0.00	nr
Yellow-billed Cuckoo	0.00-0.08	0.07-0.13	nr	0.00	Common
Yellow-throated Vireo	0.08-0.11	0.00-0.27	nr	0.04	nr
Yellow-throated Warbler	0.08-0.09	0.13-0.20	nr	0.00	nr
Total Abundance	8.53-10.47	9.69-12.25	8.00-8.99	8.28	nr
Richness	30-39	31-32	nr	43	nr

[a] PC=point count; ST=strip transect; SM=spot mapping. Actual abundance values are reported, not densities.
[b] A variation of the spot-mapping method; only relative abundance was reported.
[c] MF=mature forest; PF=pole forest.
[d] Range represents minimum and maximum values reported; single values indicate an average value; nr=abundances not reported although species do occur in that area.

Table 16. Means, standard errors (SE), and forward logistic regression results (Wald chi-square statistics) for the presence/absence of the Cerulean Warbler and Louisiana Waterthrush at point counts in forested habitats in southwestern West Virginia. The '-' and '+' indicate either a negative or a positive relationship between abundance and the habitat variables.

	Cerulean Warbler						Louisiana Waterthrush					
	Absent		Present				Absent		Present			
Variable	Mean	SE	Mean	SE	X^2	P	Mean	SE	Mean	SE	X^2	P
Aspect Code	0.98	0.08	1.17	0.13			1.03	0.08	1.15	0.16		
Slope (%)	31.75	2.02	37.28	2.15	4.08	0.04+	33.08	1.71	37.21	3.74		
Elevation (m)	376.11	9.44	361.90	14.52			376.76	8.94	341.36	15.48		
Distance to mine (m)	979.76	146.84	916.64	194.49			994.39	128.28	765.79	282.99		
Distance to closest minor edge (m)	61.98	10.52	39.11	4.73			54.74	8.27	48.07	6.52		
Canopy Height (m)	21.70	0.62	22.62	0.79			22.04	0.53	22.04	1.88		
Ground Cover (%)												
Water	0.90	0.24	0.52	0.22			0.81	0.20	0.54	0.29		
Litter	51.04	1.65	50.44	2.22			49.96	1.47	55.18	4.25		
Bareground	7.41	0.67	7.82	0.84			7.94	0.58	5.63	1.10	4.99	0.02-
Woody Debris	4.44	0.33	4.96	0.57			4.49	0.31	5.36	0.83		
Green	33.70	1.73	34.40	2.39			34.77	1.56	29.82	3.31		
Moss	2.18	0.29	1.77	0.40			1.80	0.25	3.21	0.59	6.45	0.01+
Stem Densities (no./ha)												
<2.5 cm	1826.97	99.45	1821.57	131.93			1877.20	85.95	1560.27	198.10		
>2.5-8 cm	6742.48	619.66	6990.93	781.41			7272.45	547.62	4604.91	725.28	5.28	0.02-
>8-23 cm	345.02	22.42	314.72	30.62			325.00	16.71	379.46	68.11		
>23-38 cm	96.76	4.46	88.51	6.79			94.01	4.28	92.41	8.81		
>38-53 cm	33.45	2.75	29.64	2.90			31.95	2.28	32.59	4.73		
>53-68 cm	9.61	1.31	8.87	1.61			9.60	1.10	8.04	2.49		
>68 cm	3.59	0.65	4.44	1.09			3.79	0.66	4.46	0.99		
Canopy Cover (%)												
0.5-3 m	52.31	2.14	47.90	2.81			49.63	1.82	56.16	5.59		
>3-6 m	60.28	2.01	58.79	3.05			59.96	1.83	58.57	5.49		
>6-12 m	62.73	1.56	67.42	1.93	4.19	0.04+	64.12	1.38	66.07	4.87		
>12-18 m	59.10	2.22	62.22	2.53			59.49	1.80	64.02	5.78		
>18-24 m	45.25	2.88	50.28	3.41			47.10	2.43	47.05	5.99		
>24 m	16.27	2.08	18.95	2.56			17.46	1.76	16.16	4.48		
Structural Diversity Index	11.46	0.19	11.45	0.28			11.37	0.17	11.93	0.85		

Table 17. Means, standard errors (SE), and forward logistic regression results (Wald chi-square statistics) for the presence/absence of the Worm-eating Warbler and Kentucky Warbler at point counts in forested habitats in southwestern West Virginia. The '-' and '+' indicate either a negative or a positive relationship between abundance and the habitat variables.

	Worm-eating Warbler						Kentucky Warbler					
	Absent		Present				Absent		Present			
Variable	Mean	SE	Mean	SE	X^2	P	Mean	SE	Mean	SE	X^2	P
Aspect Code	1.14	0.08	0.73	0.10	10.78	<0.01-	1.02	0.08	1.12	0.11		
Slope (%)	34.58	1.69	31.10	3.46			33.05	1.87	35.68	2.53		
Elevation (m)	374.57	8.97	359.10	17.53	2.77	0.10-	383.23	9.51	337.78	12.44	8.48	<0.01-
Distance to mine (m)	996.20	137.73	828.48	215.34			1028.68	139.65	762.82	208.64		
Distance to closest minor edge (m)	54.66	8.02	50.31	14.49			53.11	8.25	55.07	13.37		
Canopy Height (m)	21.91	0.56	22.46	1.01			21.83	0.58	22.60	0.89		
Ground Cover (%)												
Water	0.69	0.19	1.00	0.37			0.87	0.22	0.49	0.25		
Litter	51.92	1.53	47.25	2.49	3.92	0.05-	7.74	0.60	7.07	1.09		
Bareground	7.88	0.61	6.50	0.99			50.69	1.45	51.20	2.94		
Woody Debris	4.27	0.33	5.81	0.62	8.11	<0.01+	4.54	0.33	4.89	0.65		
Green	33.04	1.58	36.94	2.95			34.01	1.55	33.80	3.08		
Moss	1.98	0.28	2.19	0.43			2.00	0.28	2.12	0.44		
Stem Densities (no./ha)												
<2.5 cm	1801.44	93.74	1901.56	143.12			1908.27	95.03	1600.54	131.49	2.72	0.10-
>2.5-8 cm	6791.83	595.59	6967.19	710.55			7268.65	608.55	5658.97	665.27		
>8-23 cm	324.04	19.47	366.25	43.67			355.34	22.40	276.36	25.39	3.61	0.06-
>23-38 cm	95.29	4.61	88.75	5.60			94.46	4.26	91.85	8.00		
>38-53 cm	33.75	2.49	26.56	2.90			30.75	2.51	35.60	3.30		
>53-68 cm	9.52	1.20	8.75	1.89			9.07	1.23	10.05	1.75		
>68 cm	4.23	0.66	2.81	1.15			3.13	0.59	5.98	1.33		
Canopy Cover (%)												
0.5-3 m	51.60	1.97	47.81	3.40			51.63	2.04	48.21	3.10		
>3-6 m	59.88	1.97	59.25	3.31			59.86	1.92	59.40	3.55		
>6-12 m	64.81	1.36	63.25	2.84			63.23	1.45	67.72	2.27	4.39	<0.04+
>12-18 m	61.69	1.89	55.50	3.52	2.43	0.10-	60.52	1.90	59.46	3.62		
>18-24 m	48.92	2.41	41.13	5.15			47.30	2.55	46.52	4.55		
>24 m	16.85	1.81	18.56	3.58			17.22	1.95	17.34	2.86		
Structural Diversity Index	11.54	0.19	11.20	0.26			11.48	0.18	11.39	0.35		

Table 18. Means, standard errors (SE), and forward logistic regression results (Wald chi-square statistics) for the presence/absence of the Wood Thrush and Acadian Flycatcher at point counts in forested habitats in southwestern West Virginia. The '-' and '+' indicate either a negative or a positive relationship between abundance and the habitat variables.

	Wood Thrush						Acadian Flycatcher					
	Absent		Present				Absent		Present			
Variable	Mean	SE	Mean	SE	X^2	P	Mean	SE	Mean	SE	X^2	P
Aspect Code	1.04	0.10	1.05	0.09			0.85	0.18	1.09	0.07		
Slope (%)	31.86	2.53	35.23	1.87			33.94	3.58	33.72	1.70		
Elevation (m)	387.24	9.89	358.35	11.67	3.62	0.06-	385.06	17.80	367.65	8.94	6.70	0.01-
Distance to mine (m)	1049.47	180.64	885.26	153.19			711.22	239.19	1013.67	132.67	4.20	0.04+
Distance to closest minor edge (m)	58.52	11.58	49.88	8.63			80.72	23.55	47.36	6.53		
Canopy Height (m)	22.10	0.70	21.99	0.68			20.93	1.07	22.30	0.54		
<u>Ground Cover (%)</u>												
Water	0.71	0.29	0.81	0.21			0.94	0.61	0.72	0.16		
Litter	49.80	2.21	51.61	1.61			51.48	3.04	50.67	1.47		
Bareground	8.28	0.86	7.01	0.65			5.16	0.96	8.12	0.59	7.17	<0.01+
Woody Debris	4.80	0.46	4.51	0.39			5.47	0.85	4.44	0.31		
Green	33.99	2.48	33.93	1.60			34.77	3.00	33.77	1.58		
Moss	2.23	0.43	1.88	0.26			2.11	0.73	2.01	0.24		
<u>Stem Densities (no./ha)</u>												
<2.5 cm	1937.50	120.18	1738.28	104.05			2287.11	134.30	1717.84	87.54	3.41	0.06-
>2.5-8 cm	7456.93	760.06	6352.21	622.08			9048.83	1039.30	6319.29	528.80		
>8-23 cm	337.33	30.48	331.38	21.99			442.97	56.79	308.70	16.78	2.91	0.09-
>23-38 cm	86.15	5.93	99.61	4.72	2.98	0.08+	100.78	9.91	92.12	4.04		
>38-53 cm	32.94	3.20	31.38	2.67			34.38	5.65	31.52	2.16		
>53-68 cm	11.15	1.68	7.94	1.22			8.20	1.95	9.60	1.17		
>68 cm	4.05	0.91	3.78	0.74			1.95	0.94	4.35	0.66	1.21	0.21+
<u>Canopy Cover (%)</u>												
0.5-3 m	52.80	2.75	49.09	2.15			47.19	3.91	51.52	1.90		
>3-6 m	62.64	2.50	57.50	2.25			55.86	4.00	60.63	1.85		
>6-12 m	66.28	1.45	63.02	1.86			60.70	2.16	65.31	1.42		
>12-18 m	60.24	2.58	60.23	2.25			60.39	4.28	60.20	1.84		
>18-24 m	44.49	3.29	49.09	2.99			39.45	5.96	48.86	2.33		
>24 m	15.07	2.56	18.93	2.06			14.22	3.81	17.95	1.78		
Structural Diversity Index	11.30	0.26	11.58	0.20			10.69	0.38	11.64	0.17	3.08	0.08+

Table 19. Means, standard errors (SE), and forward logistic regression results (Wald chi-square statistics) for the presence/absence of the Hooded Warbler and Yellow-throated Vireo at point counts in forested habitats in southwestern West Virginia. The '-' and '+' indicate either a negative or a positive relationship between abundance and the habitat variables.

	Hooded Warbler						Yellow-throated Vireo					
	Absent		Present				Absent		Present			
Variable	Mean	SE	Mean	SE	χ^2	P	Mean	SE	Mean	SE	χ^2	P
Aspect Code	1.00	0.09	1.13	0.11			1.03	0.07	1.11	0.19	13.21	<0.01+
Slope (%)	33.04	2.09	34.91	2.17			32.98	1.77	36.91	2.80	5.20	0.02+
Elevation (m)	358.47	9.26	391.56	14.09			370.03	9.44	374.53	13.42	9.20	<0.01+
Distance to mine (m)	780.70	136.97	1248.30	203.05			1040.72	134.30	620.81	213.49	9.05	<0.01-
Distance to closest minor edge (m)	55.17	8.25	51.09	12.70			55.09	8.64	47.84	5.13		
Canopy Height (m)	21.25	0.67	23.28	0.63			22.40	0.56	20.59	0.88		
<u>Ground Cover (%)</u>												
Water	0.77	0.24	0.76	0.23			0.75	0.17	0.81	0.54		
Litter	8.03	0.70	6.82	0.77			7.61	0.56	7.35	1.37		
Bareground	52.16	1.74	48.71	1.97			49.83	1.38	54.78	3.53	6.46	0.01-
Woody Debris	4.30	0.33	5.15	0.55	2.61	0.10+	4.60	0.34	4.78	0.59		
Green	32.38	1.78	36.44	2.21			34.89	1.58	30.22	2.90		
Moss	2.02	0.32	2.05	0.34			2.06	0.28	1.91	0.44		
<u>Stem Densities (no./ha)</u>												
<2.5 cm	1914.66	108.99	1683.71	106.21			1779.41	83.39	2007.35	210.97		
>2.5-8 cm	6185.70	570.01	7853.22	842.86	5.19	0.02+	6784.01	563.97	7029.41	895.41		
>8-23 cm	348.68	26.89	310.80	19.03			333.64	20.65	335.29	37.69		
>23-38 cm	92.67	4.39	95.45	6.86			93.29	4.33	95.59	7.56		
>38-53 cm	31.25	2.84	33.33	2.80			30.61	2.23	37.87	4.81	2.62	0.10+
>53-68 cm	9.98	1.28	8.33	1.67			9.28	1.18	9.56	1.87		
>68 cm	3.97	0.71	3.79	0.98			3.86	0.64	4.04	1.31		
<u>Canopy Cover (%)</u>												
0.5-3 m	53.25	1.89	46.70	3.14			50.59	1.88	51.18	4.11		
>3-6 m	62.98	2.10	54.62	2.60			58.71	1.95	63.82	3.07		
>6-12 m	63.53	1.58	65.87	1.97			63.29	1.37	69.04	2.57	7.55	0.01+
>12-18 m	58.39	2.13	63.14	2.71			59.01	1.91	65.15	3.33		
>18-24 m	45.19	2.97	50.08	3.27			46.97	2.51	47.57	4.83		
>24 m	15.91	2.15	19.36	2.41			18.53	1.77	12.13	3.73		
Structural Diversity Index	11.50	0.19	11.39	0.28			11.38	0.18	11.76	0.32		

Table 20. Means, standard errors (SE), and forward logistic regression results (Wald chi-square statistics) for the presence/absence of the Black-and-white Warbler and Scarlet Tanager at point counts in forested habitats in southwestern West Virginia. The '-' and '+' indicate either a negative or a positive relationship between abundance and the habitat variables.

	Black-and-white Warbler						Scarlet Tanager					
	Absent		Present				Absent		Present			
Variable	Mean	SE	Mean	SE	χ^2	P	Mean	SE	Mean	SE	χ^2	P
Aspect Code	1.04	0.08	1.05	0.12	3.64	0.06+	1.10	0.09	0.98	0.11		
Slope (%)	32.56	2.16	35.57	2.01			30.77	1.99	37.30	2.25	8.62	<0.01+
Elevation (m)	370.14	10.18	372.12	13.03			356.13	10.31	388.38	11.99		
Distance to mine (m)	1022.10	158.37	858.70	170.12	2.95	0.09+	696.48	140.22	1263.70	182.72	9.16	<0.01+
Distance to closest minor edge (m)	58.47	9.79	46.39	9.48			59.46	12.10	46.77	5.30		
Canopy Height (m)	21.89	0.63	22.26	0.78			21.62	0.70	22.53	0.67		
<u>Ground Cover (%)</u>												
Water	0.96	0.26	0.48	0.18			0.76	0.25	0.77	0.23	3.10	0.08+
Litter	7.65	0.62	7.43	0.93			50.73	2.00	50.93	1.67		
Bareground	51.40	1.65	49.96	2.19			8.23	0.67	6.76	0.82	4.89	0.03-
Woody Debris	4.29	0.41	5.15	0.41			4.57	0.42	4.71	0.42		
Green	33.38	1.67	34.82	2.45			33.64	2.02	34.33	1.92		
Moss	2.01	0.26	2.06	0.45	6.35	0.06+	1.88	0.27	2.21	0.41		
<u>Stem Densities (no./ha)</u>												
<2.5 cm	1736.52	101.48	1957.72	123.94	10.04	<0.01-	1938.18	103.07	1691.51	119.66		
>2.5-8 cm	5866.42	474.94	8283.09	931.56	5.19	<0.01+	6770.38	482.07	6907.05	894.79		
>8-23 cm	326.96	19.73	344.49	34.44			344.43	25.99	321.63	24.91		
>23-38 cm	93.87	4.80	93.57	6.13			91.71	4.85	96.15	5.91		
>38-53 cm	32.97	2.61	30.70	3.31			29.35	2.80	35.26	2.94	6.48	0.01+
>53-68 cm	9.44	1.36	9.19	1.52			10.33	1.38	8.17	1.49		
>68 cm	3.06	0.59	5.15	1.10			3.94	0.81	3.85	0.82		
<u>Canopy Cover (%)</u>												
0.5-3 m	51.25	2.17	49.89	2.79			48.78	2.38	52.98	2.42		
>3-6 m	59.71	2.36	59.78	2.32	3.74	0.05-	59.54	2.44	59.97	2.32		
>6-12 m	62.87	1.65	66.80	1.79			63.70	1.73	65.32	1.76		
>12-18 m	59.80	2.09	60.88	2.85			55.60	2.35	65.71	2.12	6.95	<0.01+
>18-24 m	46.47	3.15	48.01	2.96			42.80	2.98	52.15	3.17		
>24 m	16.23	2.09	18.79	2.55			17.74	2.32	16.67	2.24		
Structural Diversity Index	11.43	0.20	11.50	0.26			11.24	0.21	11.72	0.24		

Table 21. Means and standard errors (SE) of songbird abundance (birds/point count) by habitat guild and nesting guild on reclaimed MTMVF areas in grassland, shrub/pole, fragmented forest, and intact forest treatments in Boone, Fayette, Kanawha, and Logan Counties, West Virginia, 1999-2000. Treatments with the same letter within rows are not significantly different (Waller-Duncan k-ratio t-test, $P \leq 0.05$).

	Treatment												ANOVA Results	
	Grassland			Shrub/pole			Fragmented Forest			Intact Forest				
Guild	1999		2000	1999		2000	1999		2000	1999		2000	F	P
Habitat														
Interior	0.20	D	0.03	1.00	C	0.36	2.67	B	3.33	4.17	A	5.70	318.66	<0.01
	(0.10)		(0.03)	(0.45)		(0.10)	(0.32)		(0.28)	(0.26)		(0.33)		
Interior-edge	0.03	D	0.33	1.50	C	2.45	3.08	A	3.33	2.58	B	2.77	182.32	<0.01
	(0.03)		(0.10)	(0.43)		(0.21)	(0.29)		(0.20)	(0.24)		(0.16)		
Edge	2.43	B	2.78	6.67	A	6.45	0.33	C	0.50	0.14	D	0.23	148.24	<0.01
	(0.39)		(0.31)	(1.48)		(0.46)	(0.12)		(0.14)	(0.07)		(0.06)		
Grass	4.33	A	4.10	0.33	B	0.67	0.00	C	0.03	0.00	C	0.00	472.39	<0.01
	(0.35)		(0.26)	(0.33)		(0.17)	(0.00)		(0.03)	(0.00)		(0.00)		
Nest														
Ground	3.60	A	3.75	2.50	B	2.27	1.46	C	1.44	1.97	C	2.11	31.88	<0.01
	(0.31)		(0.23)	(0.22)		(0.15)	(0.20)		(0.18)	(0.19)		(0.18)		
Shrub	3.27	B	3.30	5.50	A	6.27	0.42	C	0.61	0.44	C	0.64	111.27	<0.01
	(0.40)		(0.33)	(1.52)		(0.47)	(0.12)		(0.14)	(0.11)		(0.11)		
Subcanopy	0.00	C	0.13	1.67	B	0.94	3.00	A	2.42	3.06	A	2.96	204.39	<0.01
	(0.00)		(0.06)	(0.33)		(0.14)	(0.28)		(0.16)	(0.24)		(0.21)		
Canopy	0.03	B	0.00	0.00	B	0.15	0.79	A	2.17	0.92	A	2.64	1999: 15.09	1999: <0.01
	(0.03)	C	(0.00)	(0.00)	C	(0.06)	(0.16)	B	(0.21)	(0.15)	A	(0.19)	2000: 158.67	2000: <0.01
Cavity	0.00	C	0.10	0.00	B	0.76	0.88	A	1.19	0.94	A	0.87	29.70	<0.01
	(0.00)		(0.05)	(0.00)		(0.15)	(0.16)		(0.17)	(0.16)		(0.12)		
Total Abundance	8.07	C	8.28	12.17	A	12.52	7.58	BC	9.19	8.53	B	10.47	8.72	<0.01
	(0.59)		(0.41)	(1.40)		(0.59)	(0.63)		(0.51)	(0.54)		(0.47)		
Richness	5.08	C	5.17	9.36	A	9.17	7.56	B	6.71	7.91	B	7.03	22.70	<0.01
	(0.35)		(0.42)	(0.34)		(0.60)	(0.43)		(0.51)	(0.30)		(0.45)		

Table 22. Jaccard and Renkonen similarity indices comparing songbird community composition among grassland, shrub/pole, fragmented forest, and intact forest treatments in 1999 and 2000.

	Species shared		Jaccard		Renkonen	
Comparisons	1999	2000	1999	2000	1999	2000
Grassland/Intact	2	8	0.04	0.14	0.01	0.02
Grassland/Fragment	4	12	0.08	0.22	0.04	0.07
Shrub/Intact	9	21	0.20	0.37	0.17	0.12
Shrub/Fragment	11	24	0.24	0.44	0.19	0.19
Grassland/Shrub	12	23	0.40	0.48	0.33	0.42
Fragment/Intact	29	29	0.74	0.64	0.78	0.70

[a] Jaccard indices only examine the number of species shared while the Renkonen indices also take into account the proportion of each species present in each sample (in both cases the scale ranges from 0 = no similarity and 1 = complete similarity).

Table 23. Nesting success of birds on MTMVF areas by mine, nesting guild, and species.

	Year	N	Observation Days	Incubation Survival	Brooding Survival	Total Survival
Mine						
Daltex	1999	1	4.5	0.030	-----	0.030
Hobet	1999	10	66.5	0.135	0.191	0.026
Daltex	2000	13	135.5	0.546	1.000	0.546
Hobet	2000	8	88.5	0.681	1.000	0.681
Cannelton	2000	4	13.5	0.018	1.000	0.018
Combined	1999	11	71.0	0.160	0.258	0.041
Combined	2000	25	237.5	0.527	1.000	0.527
Nesting Guilds						
Shrub	1999	2	11.5	0.101	-----	0.101
Ground	1999	8	52.5	0.166	0.222	0.037
Shrub	2000	3	54.0	1.000	1.000	1.000
Ground	2000	18	158.0	0.329	1.000	0.329
Miscellaneous[a]	2000	2	20.0	1.000	1.000	1.000
Years Combined						
Shrub	99/00	5	65.5	0.488	1.000	0.488
Ground	99/00	26	210.5	0.262	0.774	0.203
Miscellaneous	99/00	2	20.0	1.000	1.000	1.000
Species						
Barn Swallow	99/00	1	4.5	-----	1.000	1.000
Eastern Bluebird	99/00	1	15.5	1.000	1.000	1.000
Eastern Meadowlark	99/00	1	16.0	1.000	1.000	1.000
Field Sparrow	99/00	2	12.0	0.180	0.134	0.024
Grasshopper Sparrow	99/00	19	172.0	0.397	0.917	0.364
Horned Lark	99/00	2	4.5	0.008	0.000	0.000
Indigo Bunting	99/00	2	11.5	0.083	-----	0.083
Killdeer	99/00	3	35.0	0.230	-----	-----
Mourning Dove	99/00	2	6.0	0.003	-----	0.003
Red-winged Blackbird	99/00	3	54.0	1.000	1.000	1.000

[a] Eastern Bluebird and Barn Swallow.

Table 24. Comparison of grassland bird nest survival on reclaimed MTMVF areas with previous studies.

Species	No. Nests (years)	Nest Density[b]	Nest Survival	Location	Grassland Type[a]	Study
Grasshopper Sparrow	19 (2)	~0.06/ha	0.36	West Virginia	MTMVF	This study
	51 (3)	0.11/ha	0.07	West Virginia	Surface mines	Wray (1982)
	38 (3)	nr	0.41	Missouri	CRP field- warm/cool season grasses	McCoy et al. (1999)
	12 (1)	0.06/ha	0.41	Illinois	Airport grasslands	Kershner & Bollinger (1996)
	14 (3)	nr	0.11	North Dakota	WPA	Koford (1999)
	38 (3)	nr	0.28	North Dakota	CRP fields	Koford (1999)
	13 (3)	nr	0.12	Minnesota	CRP fields	Koford (1999)
	12 (3)	0.25/ha	~0.25[c]	Oklahoma	Tallgrass prairie	Rohrbaugh et al. (1999)
Eastern Meadowlark	1 (2)	<0.01/ha	1.00	West Virginia	MTMVF	This study
	12 (3)	nr	0.67	New York	Pasture/cool season grass	Norment et al. (1999)
	32 (3)	nr	0.30	Missouri	CRP fields- warm/cool season grasses	McCoy et al. (1999)
	105 (1)	0.56/ha	0.14	Illinois	Airport grasslands	Kershner & Bollinger (1996)
	42 (3)	0.86/ha	~0.25[b]	Oklahoma	Undisturbed tallgrass prairie	Rohrbaugh et al. (1999)
	7 (1)	nr	0.62	West Virginia	Pastures/wet meadows	Warren & Anderson, (unpub. data)
Horned Lark	2 (2)	~0.01/ha	0.00	West Virginia	MTMVF	This study
	47 (2)	0.23/ha	0.05	West Virginia	Surface mines	Wackenhut (1980)
	3 (1)	0.02/ha	1.00	Illinois	Airport grasslands	Kershner & Bollinger (1996)
Red-winged Blackbird	3 (2)	~0.01/ha	1.00	West Virginia	MTMVF	This study
	145 (6)	1.41/ha	0.48	Illinois	Cool season grasslands	Warner (1994)
	70 (3)	nr	0.11	North Dakota	CRP fields	Koford (1999)
	9 (3)	nr	0.17	North Dakota	WPA	Koford (1999)
	25 (3)	nr	0.01	Minnesota	CRP fields	Koford (1999)
	63 (2)	5.66/ha	0.08	Iowa	Grassed waterways	Bryan & Best (1994)
	238 (3)	nr	0.28	Missouri	CRP fields - warm/cool season grasses	McCoy et al. (1999)
	11 (1)	0.06/ha	0.06	Illinois	Airport grasslands	Kershner & Bollinger (1996)
	15 (1)	nr	0.42	West Virginia	Pastures/wet meadows	Warren & Anderson, (unpub. data)

Table 24. Continued.

Species	No. Nests (years)	Nest Density	Nest Survival	Location	Grassland Type[a]	Study
Savannah Sparrow	0 (2)	----	----	West Virginia	MTMVF	This study
	41(3)	0.24/ha	0.22	West Virginia	Surface mines	Wray (1982)
	58 (3)	nr	0.76	New York	Pasture/cool season grass	Norment et al. (1999)
	12 (1)	0.02/ha	0.23	Illinois	Airport grasslands	Kershner & Bollinger (1996)
	4 (3)	nr	0.15	North Dakota	CRP fields	Koford (1999)
	4 (3)	nr	0.22	North Dakota	WPA	Koford (1999)
	12 (3)	nr	0.02	Minnesota	CRP fields	Koford (1999)
	30 (3)	nr	0.25	Minnesota	WPA	Koford (1999)
	17 (1)	nr	0.36	West Virginia	Pastures/wet meadows	Warren & Anderson, (unpub. data)
Dickcissel	0 (2)	----	----	West Virginia	MTMVF	This study
	14 (6)	0.14/ha	0.14	Illinois	Cool season grassland	Warner (1994)
	27 (2)	2.76/ha	0.22	Iowa	Grassed waterways	Bryan & Best (1994)
	87 (3)	nr	0.30	Missouri	CRP field- warm/cool season grasses	McCoy et al. (1999)
	87 (3)	0.60/ha	~0.25[b]	Oklahoma	Tallgrass prairie	Rohrbaugh et al. (1999)

[a] MTMVF = mountaintop mining/valley fill; CRP = conservation reserve program; WPA = waterfowl production area.
[b] nr=not reported.
[c]Survival rates were presented in a figure and estimates are approximate.

Table 25. Means and standard errors of habitat variables surrounding successful (n=11) and unsuccessful (n=4) nests of Grasshopper Sparrows on MTMVF areas in 2000.

	Successful		Unsuccessful		Combined	
Variable	Mean	SE	Mean	SE	Mean	SE
Aspect Code	1.5	0.4	3.2	0.8	2.0	0.4
Slope (%)	4.8	2.2	16.0	10.0	7.8	3.2
Overhead Cover (%)	47.5	10.3	28.8	10.5	42.5	8.2
Side Cover (%)	85.1	4.7	74.3	23.3	82.2	6.6
Distance to Minor Edge (m)	22.8	7.4	36.3	3.8	26.4	5.7
Lespedeza Cover (%)	5.8	3.7	0.3	0.3	4.3	2.7
Ground Cover (%)						
Green	81.4	4.1	88.8	9.7	83.3	3.9
Grass	43.2	6.0	47.5	8.3	44.3	4.8
Forb	35.9	5.6	37.5	14.5	36.3	5.3
Shrub	2.3	1.0	0.0	0.0	1.7	0.8
Litter	0.0	0.0	0.0	0.0	0.0	0.0
Wood	0.0	0.0	0.0	0.0	0.0	0.0
Bareground	18.6	4.1	6.3	4.7	15.3	3.5
Moss	0.0	0.0	5.0	5.0	1.3	1.3
Water	0.0	0.0	0.0	0.0	0.0	0.0
Robel Pole Index						
nest	2.5	0.3	2.2	0.4	2.4	0.2
1m	2.5	0.3	2.6	0.3	2.5	0.2
3m	2.7	0.2	2.2	0.4	2.6	0.2
5m	2.3	0.2	2.6	0.4	2.4	0.2
Grass Height (dm)						
nest	4.6	1.0	3.8	1.4	4.4	0.8
1m	5.3	0.6	5.3	0.9	5.3	0.5
3m	5.5	0.5	6.2	0.8	5.7	0.4
5m	4.8	0.5	7.2	0.3	5.5	0.4
10m	5.3	0.3	7.6	0.6	5.9	0.4
Litter depth (cm)						
nest	2.0	0.3	0.5	0.3	1.6	0.3
1m	2.0	0.5	1.3	0.4	1.8	0.4
3m	1.8	0.4	0.8	0.2	1.5	0.3
5m	2.2	0.4	1.8	0.3	2.1	0.3

Table 26. Seasonal mean abundance (no./survey), species richness, and standard errors (SE) of raptors during broadcast surveys across in grassland, shrub/pole, fragmented forest, and intact forest treatments on reclaimed MTMVF areas in 2000.

	Grassland						Shrub/pole					
	Winter		Summer		Migration		Winter		Summer		Migration	
Species	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Overall Abundance	0.33	0.25	1.10	0.30	0.67	0.16	0.21	0.12	0.23	0.10	0.46	0.27
Overall Richness	0.08	0.06	0.08	0.04	0.10	0.04	0.17	0.10	0.06	0.05	0.06	0.04
American Kestrel	0.00	0.00	0.04	0.03	0.13	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Peregrine Falcon	0.00	0.00	0.00	0.00	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00
Cooper's Hawk	0.00	0.00	0.00	0.00	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00
Accipiter spp.[a]	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.00	0.00	0.00
Northern Harrier	0.08	0.06	0.04	0.03	0.13	0.05	0.00	0.00	0.02	0.02	0.00	0.00
Red-tailed Hawk	0.00	0.00	0.08	0.04	0.06	0.04	0.04	0.04	0.02	0.02	0.00	0.00
Red-shouldered Hawk	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.00	0.04	0.03
Eastern Screech Owl	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.02
Barred Owl	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Turkey Vulture	0.25	0.25	0.94	0.29	0.31	0.14	0.08	0.08	0.19	0.09	0.44	0.27
Unknown	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

[a] Either Sharp-shinned Hawk or Cooper's Hawk.

Table 26. Continued.

Species	Fragmented Forest						Intact Forest					
	Winter		Summer		Migration		Winter		Summer		Migration	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Overall Abundance	0.21	0.10	0.21	0.08	0.13	0.06	0.25	0.12	0.17	0.08	0.16	0.07
Overall Richness	0.08	0.06	0.06	0.04	0.06	0.05	0.13	0.09	0.08	0.04	0.05	0.04
American Kestrel	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Peregrine Falcon	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cooper's Hawk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Accipiter spp.[a]	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Northern Harrier	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Red-tailed Hawk	0.17	0.10	0.06	0.03	0.02	0.02	0.04	0.04	0.02	0.02	0.05	0.04
Red-shouldered Hawk	0.00	0.00	0.02	0.02	0.06	0.04	0.04	0.04	0.08	0.05	0.11	0.07
Eastern Screech Owl	0.00	0.00	0.00	0.00	0.04	0.03	0.00	0.00	0.00	0.00	0.00	0.00
Barred Owl	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.02	0.00	0.00
Turkey Vulture	0.04	0.04	0.10	0.07	0.00	0.00	0.17	0.10	0.02	0.02	0.00	0.00
Unknown	0.00	0.00	0.02	0.02	0.00	0.00	0.00	0.00	0.02	0.02	0.00	0.00

[a] Either Sharp-shinned Hawk or Cooper's Hawk.

Table 27. Abundance and richness of raptor species observed on roadside surveys in grassland, shrub/pole, and fragmented forest treatments on each of the 3 MTMVF areas in 2000.

	Hobet			Cannelton			Daltex	
Species	Grass	Shrub/ pole	Fragmented Forest	Grass	Shrub/ pole	Fragmented Forest	Grass	Fragmented Forest
Overall Abundance	11	7	2	2	1	7	14	11
Overall Richness	3	2	1	2	1	3	4	1
American Kestrel	3	0	0	0	0	0	5	0
Peregrine Falcon	0	0	0	0	0	0	1	0
Northern Harrier	0	0	0	1	1	0	0	0
Broad-winged Hawk	0	0	0	0	0	1	0	0
Red-tailed Hawk	2	1	0	1	0	4	1	0
Turkey Vulture	6	6	2	0	0	2	6	11
Unknown[a]	0	0	0	0	0	0	1	0

[a] Unknown is either a Red-tailed Hawk or Turkey Vulture.

Table 28. Seasonal observations of raptor species (w=winter, s=summer, m=migration) on the 3 mines in each of the 4 treatments (GR=grassland, SH=shrub/pole, FR=fragmented forest, IN=intact forest), compared to species expected based on habitat requirements and West Virginia Breeding Bird Atlas (WV BBA) records.

	WV BBA	Expected in WV from habitat requirements[a]				Observations on the 3 mines[b]										
						Hobet				Daltex			Cannelton			
Species	Record	GR	SH	FR	IN	GR	SH	FR	IN	GR	FR	IN	GR	SH	FR	IN
American Kestrel	s	wsm	wsm	wm	wm	wsm		m		sm			s	sm		
Peregrine Falcon				m	m					sm			s[c]m			
Northern Harrier		m				wsm				wm			wsm	sm		
Broad-winged Hawk	s				sm	s						s			sm	
Red-shouldered Hawk	s				wsm	s	wsm	s	sm	s	m	s		ws	sm	wsm
Red-tailed Hawk	s	s	s	wsm	wsm	sm	wsm	wsm	wsm	sm	ws	sm	sm	wsm	sm	
Rough-legged Hawk				wm	wm	w		w								
Cooper's Hawk	s	s	s	sm	sm	m				m		s	m			
Sharp-shinned Hawk	s		s	sm	sm	s								w		
Accipiter spp.[d]	s	s	s	sm	sm			s			w					
Barred Owl	s		s	wsm	wsm			s	s			s		w		
Eastern Screech Owl	s		s	wsm	wsm		m	s			sm					
Short-eared Owl		wm								w						
Turkey Vulture	s	wsm	wsm	wsm	s	sm	wsm	s	wsm	sm	ws		wsm	sm	wsm	

[a]Buckelew and Hall (1994), Hall (1983), and West Virginia GAP analysis data.
[b]Includes observations from broadcast surveys and roadside surveys in 2000, and incidental sightings for 1999 and 2000
[c]Unconfirmed sighting.
[d]Either Sharp-shinned Hawk or Cooper's Hawk.

Table 29. Similarity indices comparing raptor community composition among treatments for all seasons in 2000.

Comparison	Species shared	Jaccard[a]	Renkonen
Grassland/Intact	2	0.25	0.08
Grassland/Fragment	2	0.25	0.11
Fragment/Intact	3	0.60	0.12
Shrub/Intact	3	0.43	0.09
Shrub/Fragment	4	0.67	0.10
Shrub/Grassland	3	0.33	0.29

[a]The Jaccard index only examines the number of species shared while the Renkonen index takes into account the proportion of each species present in each sample (in all cases the scale ranges from 0=no similarity and 1=complete similarity).

Table 30. Mammal species expected (Exp) to occur in grassland, shrub/pole, fragmented forest and intact forest treatments and reclaimed-mine ponds based on WV GAP analysis data, personal communication by M. E. Hight (2000), and Whitaker and Hamilton (1998) compared to species observed (Obs) via several methods including Sherman live trapping (s), pitfall trapping (p), and incidental sighting (i).

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		Pond[a]	
Species	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs
Order Insectivora										
Hairy-tailed mole *Parascalops breweri*			x		x	i	x			
Masked Shrew *Sorex cinereus*	x	p, s	x	p	x	p, s	x	p		
Pygmy shrew *Sorex hoyi*	x	p	x	p	x	p	x	p		
Short-tailed shrew *Blarina brevicauda*	x	p, s	x	p	x	p, s	x	p, s	x	
Smoky shrew *Sorex fumeus*		p	x	p	x	p	x	p		
Order Rodentia										
Allegheny woodrat *Neotoma magister*				s			x			
Beaver *Castor canadensis*			x		x	i	x		x	
Eastern chipmunk *Tamias striatus*		s	x		x	s	x	s		
Eastern fox squirrel *Sciurus niger*	x				x		x			
Eastern gray squirrel *Sciurus carolinensis*	x				x		x			
Golden mouse *Ochrotomys nuttalli*			x		x		x			
Groundhog *Marmota monax*	x	i	x		x		x			
House mouse *Mus musculus*	x	p, s								s
Meadow vole *Microtus pennsylvanicus*	x	p, s	x	p, s	x	p	x			
Muskrat *Ondatra zibethicus*	x				x		x		x	

Table 30. Continued.

Species	Grassland Exp	Grassland Obs	Shrub/pole Exp	Shrub/pole Obs	Fragmented Forest Exp	Fragmented Forest Obs	Intact Forest Exp	Intact Forest Obs	Pond[a] Exp	Pond[a] Obs
Peromyscus species *P. leucopus/maniculatus*	x	p, s	x	p, s	x	p, s	x	p, s		s
Red squirrel *Tamiasciurus hudsonicus*			x		x		x			
Southern bog lemming *Synaptomys cooperi*	x	p, s	x	p, s	x	p	x	p	x	s
Southern flying squirrel *Glaucomys volans*			x		x		x			
Southern red-backed vole *Clethrionomys gapperi*			x		x		x			
Woodland jumping mouse *Napaeozapus insignis*		p	x		x	p, s	x	s		
Woodland vole *Microtus pinetorum*			x	p	x	p	x	s		
Order Carnivora										
Black bear *Ursus americanus*	x	i	x	i	x	i	x	i	x	
Bobcat *Lynx rufus*			x		x	i	x		x	
Coyote *Canis latrans*	x	i	x	i	x	i	x		x	
Gray fox *Urocyon cinereoargenteus*	x		x		x		x		x	
Least weasel *Mustela nivalis*			x		x		x			
Long-tailed weasel *Mustela frenata*			x		x		x			
Mink *Mustela vison*					x		x			
Raccoon *Procyon lotor*	x		x	i	x	i	x	i	x	i
Red fox *Vulpes vulpes*	x	i	x		x	i	x		x	
Striped skunk *Mephitis mephitis*	x		x	i	x		x			

Table 30. Continued.

	Treatment									
	Grassland		Shrub/pole		Fragment ed Forest		Intact Forest		Pond[a]	
Species	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs
Other										
Eastern cottontail *Sylvilagus floridanus*	x	s, i	x	i	x	i	x	i	x	i
Virginia opossum *Didelphis virginiana*			x		x	i	x	i	x	
White-tailed deer *Odocoileus virginianus*	x	i	x	i	x	i	x	i	x	i
Wild boar *Sus scrofa*			x		x		x			i

[a] Ponds were not considered a treatment because they were distributed throughout the reclaimed areas, overlapping both grassland and shrub/pole treatments.

Table 31. Average mammalian species richness (# species/transect), relative abundance (mammals/100 trap nights), and standard errors (SE) in grassland,shrub/pole, fragmented forest, and intact forest treatments and reclaimed-mine ponds in 1999 and 2000. Means were compared among treatments within years; means followed by different letters are significantly different (*P*=0.05) from each other. An absence of letters beside the means indicates that they were not subjected to statistical analysis due to small sample size.

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		Pond[a]	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Species Richness										
1999	1.7 A (n=16)[b]	0.18	-[c]	-	1.8 A (n=16)	0.25	2.3 A (n=16)	0.19	-	-
2000	1.4 A (n=20)	0.13	1.5 A (n=12)	0.15	1.4 A (n=20)	0.15	1.4 A (n=20)	0.13	1.1 (n=56)	0.09
Relative Abundance										
Total										
1999	16.1 A	1.66	-	-	12.6 A	0.94	14.5 A	1.87	-	-
2000	21.8 A	2.38	20.2 A	2.74	7.5 B	1.07	7.9 B	1.83	8.9	1.05
Peromyscus species										
1999	13.9 A	1.30	-	-	10.8 A	0.69	11.3 A	1.59	-	-
2000	20.4 A	2.58	18.9 A	2.52	6.0 B	0.78	6.6 B	1.66	7.8	1.02
House mouse										
1999	1.9	0.83	-	-	0.0	0.00	0.0	0.00	-	-
2000	1.0	0.59	0.0	0.00	0.0	0.00	0.0	0.00	0.5	0.22
Woodland jumping mouse										
1999	0.0	0.00	-	-	0.7	0.39	0.0	0.00	-	-
2000	0.0	0.00	0.0	0.00	1.0	0.58	0.5	0.27	0.0	0.00
Meadow vole										
1999	0.1	0.08	-	-	0.0	0.00	0.0	0.00	-	-
2000	0.0	0.00	0.3	0.17	0.0	0.00	0.0	0.00	0.1	0.06
Short-tailed shrew										
1999	0.3 A	0.27	-	-	0.9 AB	0.38	2.1 B	0.62	-	-
2000	0.0	0.00	0.0	0.00	0.2	0.12	0.0	0.00	0.0	0.00
Eastern chipmunk										
1999	0.0 A	0.00	-	-	0.1 A	0.08	0.9 B	0.31	-	-
2000	0.1 A	0.07	0.0 A	0.00	0.1 A	0.06	0.8 B	0.35	0.0	0.00
Eastern woodrat										
1999	0.0	0.00	-	-	0.0	0.00	0.0	0.00	-	-
2000	0.0	0.00	1.2	0.67	0.0	0.00	0.0	0.00	0.1	0.09
Southern bog lemming										
1999	0.0	0.00	-	-	0.0	0.00	0.0	0.00	-	-
2000	0.1	0.09	0.1	0.10	0.0	0.00	0.0	0.00	0.3	0.13

Table 31. Continued.

	Treatment									
	Grassland		Shrub/pole		Fragmented Forest		Intact Forest		Pond[a]	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Relative Abundance										
Masked shrew										
1999	0.0	0.00	-	-	0.1	0.08	0.1	0.10	-	-
2000	0.0	0.00	0.0	0.00	0.1	0.06	0.0	0.00	0.0	0.00
Virginia Opossum										
1999	0.0	0.00	-	-	0.3	0.30	0.0	0.00	-	-
2000	0.1	0.09	0.0	0.00	0.0	0.00	0.0	0.00	0.0	0.00
Eastern cottontail										
1999	0.1	0.06	-	-	0.0	0.00	0.0	0.00	-	-
2000	0.3	0.20	0.0	0.00	0.0	0.00	0.0	0.00	0.0	0.00

[a] Data were not included in the statistical analysis because the trapping methods were different from those used in the other three treatments.
[b] n= the number of “surveys” where a “survey” is a single transect trapped for 3 nights (or 2 nights for ponds).
[c] The shrub/pole treatment and ponds were not sampled in 1999.

Table 32. Similarity indices comparing small mammal community composition among treatments in 1999 and 2000.

	Species Shared		Jaccard[a]		Renkonen	
Comparison	1999	2000	1999	2000	1999	2000
Grassland/Intact	2	2	0.25	0.29	0.79	0.83
Grassland/Fragment	2	2	0.22	0.22	0.86	0.81
Fragment/Intact	4	2	0.57	0.33	0.87	0.86
Shrub/Intact	-[b]	1	-	0.17	-	0.83
Shrub/Fragment	-	1	-	0.13	-	0.80
Shrub/Grassland	-	2	-	0.25	-	0.93

[a] The Jaccard index only examines the number of species shared while the Renkonen index takes into account the proportion of each species present in each sample (in all cases the scale ranges from 0 = no similarity to 1 = complete similarity).

[b] A dash indicates that comparisons were not possible since “Shrub” treatment was not sampled in 1999.

Table 33. A comparison of the small mammal abundances found on our study with those of other studies. These comparisons should be interepreted with caution, however, because none occurred on MTMVF areas and sampling methods differed.

Study	Location	Duration	Study Area	Trap Type	Years Since Reclamation	Correction[a] Employed?	Abundance (per 100 trap nights) Total	*Peromyscus*[b] species	House mouse	Meadow vole	Short-tailed shrew
Grassland Studies											
Our study	Southern W. Va.	1999-2000	MTM[c]	Live	5-15	Yes	18.9	17.1	1.4	0.1	0.1
Our study	Southern W. Va.	1999-2000	MTM[c]	Live	5-15	No	13.4	12.0	1.2	0.0	0.1
Verts (1957)	Southern Ill.	1954	SM[c]	Snap	4-15	No	nr[d]	14.7	nr[d]	nr[d]	nr[d]
Voight and Glenn-Lewin (1979)	Southern Iowa	1975-1976	SM[c]	Snap	14-24	No	12.6	10.9	0.0	0.5	0.2
Mindell (1978)	Northern W. Va.	1977-1978	SM[c]	Snap	2-6	No	5.1	0.7	0.1	4.1	0.2
Forren (1981)	Northern W Va.	1980	SM[c]	Snap	4-9	No	4.1	0.2	nr[d]	2.3	1.5
Sly (1976)	Ind.	1969	SM[c]	Snap	5-12	No	6.0	5.3	0.05	0.05	0.1
Kirkland (1976)	Central New York	1973	SM[c]	Live	1-20	No	3.2	2.7	0.0	0.02	0.02
Clark et al. (1998)	Southeastern Okl.	1991	GR[c]	Snap	na[e]	No	16.9	3.7	1.6	nr[d]	0.1
Sietman et al. (1994)	East-central Kan.	1991	GR[c]	Live	na[e]	No	4.8	1.9	0.0	0.0	0.0
Denmon (1998)	W. Va.	1996-1997	ES[c]	Snap	5-20	Yes	2.7	1.0	0.0	0.3	0.7
Shrub/pole Studies											
Our study	Southern W. Va.	2000	MTM[c]	Live	16-32	Yes	20.2	18.9	0.0	0.3	0.0
Our study	Southern W. Va.	2000	MTM[c]	Live	16-32	No	14.1	13.2	0.0	0.2	0.0
Verts (1957)	Southern Ill.	1954	SM[c]	Snap	16-22	No	nr[d]	7.6	nr[d]	nr[d]	nr[d]
Denmon (1998)	W. Va.	1996-1997	ES[c]	Snap	21-30	Yes	3.4	2.7	0.0	0.3	0.2

[a] Refers to correction for sprung traps used in abundance calculations. One-half a trap night is subtracted for each sprung trap in order to more accurately reflect trapping effort (Nelson and Clark 1973). We calculated our abundances with and without the correction since some of the studies to which we compared our results employed the correction while some did not. We assumed that other studies did not employ a correction if they did not state in their methods that they did so.

[b] Includes white-footed mice (*Peromyscus leucopus*) and deer mice (*Peromyscus maniculatus*)

[c] MTM = Reclaimed mountaintop mine/valleyfill
SM = Reclaimed strip mine
GR = Natural grassland
ES = Land in early successional stage following mining or logging operations.

[d] nr = Value not reported

[e] na = Not applicable

Table 34. Species expected (Exp) to occur on in grassland, shrub/pole, fragmented forest, and intact forest treatments in our study area in southwestern West Virgini,a based on Green and Pauley (1987) and personal communication with T. Pauley, compared to those actually observed (Obs) in drift fence surveys (a), stream searches (s), and from incidental sightings (i).

	Grassland		Shrub/ pole		Fragmented Forest		Intact Forest	
Species	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs
<u>Terrestrial species</u>								
Salamanders								
Cumberland Plateau Salamander (*Plethodon kentucki*)					x		x	a
Four-toed Salamander (*Hemidactylium scutatum*)					x		x	
Green Salamander (*Aneides aeneus*)					x		x	
Jefferson Salamander (*Ambystoma jeffersonianum*)					x		x	
Longtail Salamander (*Eurycea longicauda*)	x		x		x	a	x	
Marbled Salamander (*Ambystoma opacum*)					x		x	
Ravine Salamander (*Plethodon richmondi*)					x		x	
Redback Salamander (*Plethodon cinereus*)					x		x	a
Red Eft (*Notophthalmus viridescens*) [a]		a		a	x	a	x	a
Slimy Salamander (*Plethodon glutinosus*)					x	a	x	a
Spotted Salamander (*Ambystoma maculatum*)					x	a	x	
Wehrle's Salamander (*Plethodon wehrlei*)					x		x	
Toads and frogs								
Eastern American Toad (*Bufo americanus*)	x	a	x	a		a		a
Eastern Spadefoot (*Scaphiopus holbrookii*)					x		x	
Fowler's Toad (*Bufo woodhouseii*)			x					
Gray Treefrog (*Hyla chrysoscelis*)				i	x		x	
Mountain Chorus Frog (*Pseudacris brachyphona*)					x		x	
Northern Peeper (*Pseudacris crucifer*)					x	a	x	
Wood Frog (*Rana sylvatica*)					x		x	a
Lizards								
Broadhead Skink (*Eumeces laticeps*)					x		x	
Five-lined Skink (*Eumeces fasciatus*)	x		x	a	x	a	x	a
Ground Skink (*Scincella lateralis*)					x		x	a
Northern Coal Skink (*Eumeces anthracinus*)	x		x		x		x	
Northern Fence lizard (*Sceloporus undulatus*)	x	a		a		i		
Snakes								
Black King Snake (*Lampropeltis getulus*)	x		x		x		x	
Black Rat Snake (*Elaphe obsoleta*)	x	a	x	a	x	a	x	i
Eastern Earth Snake (*Virginia valeriae*)	x		x		x		x	
Eastern Garter Snake (*Thamnophis sirtalis*)	x	a	x	a	x	a	x	a
Eastern Hognose (*Heterodon platirhinos*)	x	a		a				
Eastern Milk Snake (*Lampropeltis triangulum*)	x		x	a	x	a	x	a
Eastern Smooth Green Snake (*Opheodrys vernalis*)	x			i				i
Eastern Worm Snake (*Carphophis amoenus*)	x		x		x		x	a
Northern Black Racer (*Coluber constrictor*)	x	a	x	a		i		i
Northern Brown Snake (*Storeria dekayi*)	x		x		x		x	
Northern Copperhead (*Agkistrodon contortrix*)				a	x	a	x	a
Northern Redbelly Snake (*Storeria occipitomaculata*)	x		x		x	a	x	a
Northern Ringneck Snake (*Diadophis punctatus*)					x		x	i
Rough Green Snake (*Opheodrys aestivus*)	x		x		x		x	

Table 34. Continued.

Species	Grassland		Shrub/ pole		Fragmented Forest		Intact Forest	
	Exp	Obs	Exp	Obs	Exp	Obs	Exp	Obs
Timber Rattlesnake (*Crotalus horridus*)				i	x	l	x	
Turtles								
Eastern Box Turtle (*Terrapene carolina*)	x		x	a	x	a	x	a
<u>**Aquatic species**</u>								
Salamanders								
Appalachian Seal Salamander (*Desmognathus monticola*)					x	s	x	a
Dusky Salamander spp. (*D.fuscus* or *D.ochrophaeus*)					x		x	
Eastern Hellbender (*Cryptobranchus alleganiensis*)					x		x	
Midland Mud Salamander (*Pseudotriton montanus*)					x		x	
Mudpuppy (*Necturus maculosus*)	x		x		x		x	
Northern Dusky Salamander (*Desmognathus fuscus*)					x	s	x	s
Northern Red Salamander (*Pseudotriton ruber*)	x		x		x		x	
Red-spotted Newt (*Notophthalmus viridescens*)	x	a	x	a	x	a	x	a
Southern Two-lined Salamander (*Eurycea cirrigera*)					x		x	i
Spring Salamander (*Gyrinophilus porphyriticus*)					x		x	i
Toads and frogs								
Bullfrog (*Rana catesbeiana*)	x		x	a	x	a	x	
Green Frog (*Rana clamitans*)	x	a	x	a	x	a	x	a
Northern Leopard Frog (*Rana pipiens*)	x		x		x		x	
Pickerel frog (*Rana palustris*)	x	a	x	a	x	a	x	a
Snakes								
Northern Water Snake (*Nerodia sipedon*)	x	a	x	a	x	i	x	
Queen Snake (*Regina septemvittata*)					x		x	
Turtles								
Common Snapping Turtle (*Chelydra serpentina*)	x	i	x	i	x	i	x	
Eastern Spiny Softshell Turtle (*Trionyx spiniferus*)	x		x	i	x		x	
Midland Painted Turtle (*Chrysemys picta*)	x		x		x		x	
Stinkpot (*Sternotherus odoratus*)	x		x		x		x	

[a] Juvenile form of red-spotted newt; not included as a separate species in calculations of species richness.

Table 35. Herpetofaunal species richness and relative abundance in grassland, shrub/pole, fragmented forest, and intact forest treatments on reclaimed MTMVF areas in southwestern West Virginia, March - September, 2000.

	Treatment			
	Grassland	Shrub/pole	Fragmented Forest	Intact Forest
Species Richness				
No. species	13	14	16	15
Mean	0.21	0.28	0.29	0.22
SE	0.04	0.03	0.04	0.03
Overall Abundance				
No. individuals	91	109	110	59
Mean	0.52	0.61	0.63	0.34
SE	0.19	0.15	0.13	0.06

Table 36. Herpetofaunal community similarity between pairs of treatments on reclaimed MTMVF areas in southwestern West Virginia, March - September, 2000.

Comparisons	No. species shared	Jaccard[a] index	Renkonen index
Grassland/Shrub	11	0.69	0.65
Grassland/Fragment	9	0.45	0.58
Grassland/Intact	6	0.27	0.43
Shrub/Fragment	10	0.50	0.55
Shrub/Intact	7	0.32	0.56
Fragment/Intact	10	0.48	0.61

[a]The Jaccard index only examines the number of species shared while the Renkonen index takes into account the proportion of each species present in each sample (in all cases the scale ranges from 0=no similarity and 1=complete similarity).

Table 37. Number of individuals and species of herpetofauna groups captured in drift fence arrays in grassland, shrub/pole, fragmented forest, and intact forest treatments on reclaimed MTMVF areas in southwestern West Virginia, March - September, 2000.

	Grassland				Shrub/pole				Fragmented Forest				Intact Forest			
	Individuals		Species		Individuals		Species		Individuals		Species		Individuals		Species	
Taxonomic Group	n	%	n	%	n	%	n	%	n	%	n	%	n	%	n	%
Salamanders	5	5.7	2	15.4	5	4.6	1	7.1	25	23.1	4	25.0	17	29.3	4	26.7
Toads and frogs	63	71.6	3	23.1	68	63.0	4	28.6	65	60.2	5	31.3	31	53.4	4	26.7
Lizards	2	2.3	1	7.7	2	1.9	2	14.3	3	2.8	1	6.3	2	3.4	2	13.3
Snakes	17	19.3	6	46.2	33	30.6	7	50.0	13	12.0	5	31.3	6	10.3	4	26.7
Turtles	1	1.1	1	7.7	0	0.0	0	0.0	2	1.9	1	6.3	2	3.4	1	6.7

Table 38. Number of individuals (# indivs) of herpetofauna species captured in drift fence arrays and percent of points at which a species was captured in grassland, shrub/pole, fragmented forest, and intact forest treatments on reclaimed MTMVF areas in southwestern West Virginia, March - September, 2000.

	Grassland		Shrub/pole		Fragmented Forest		Intact Forest	
Species	# indivs	% of points	# indivs	% of points	# indivs	% of points	# indivs	% of points
Salamanders								
Appalachian Seal Salamander							1	33
Cumberland Plateau Salamander							4	66
Longtail Salamander					2	33		
Redback Salamander							2	33
Red-spotted Newt	4	100	5	100	19	100	10	100
Slimy Salamander					3	33		
Spotted Salamander	1	33			1	33		
Toads and frogs								
Bullfrog			2	66	1	33		
Eastern American Toad	7	66	27	100	3	66	14	100
Green Frog	39	100	25	100	26	66	4	100
Northern Spring Peeper					3	66		
Pickerel Frog	17	100	14	66	32	100	12	66
Unidentified Frog	2	33	1	33			1	33
Wood Frog							1	33
Lizards								
Five-lined Skink			1	33	3	33	1	33
Ground Skink							1	33
Northern Fence Lizard	2	66	1	33				
Snakes								
Black Rat Snake	6	66	4	66	1	33		
Eastern Garter Snake	3	33	5	66	7	66	2	33
Eastern Hognose	1	33	1	33				
Eastern Milk Snake	1	33	2	33	1	33		
Eastern Worm Snake							1	33
Northern Black Racer	5	66	14	100				
Northern Copperhead			6	66	3	66	2	66
Northern Redbelly Snake					1	33	1	33
Northern Water Snake	1	33	1	33				
Turtles								
Eastern Box Turtle	1	33			2	66	2	33
Unknown	1	33			2	33		

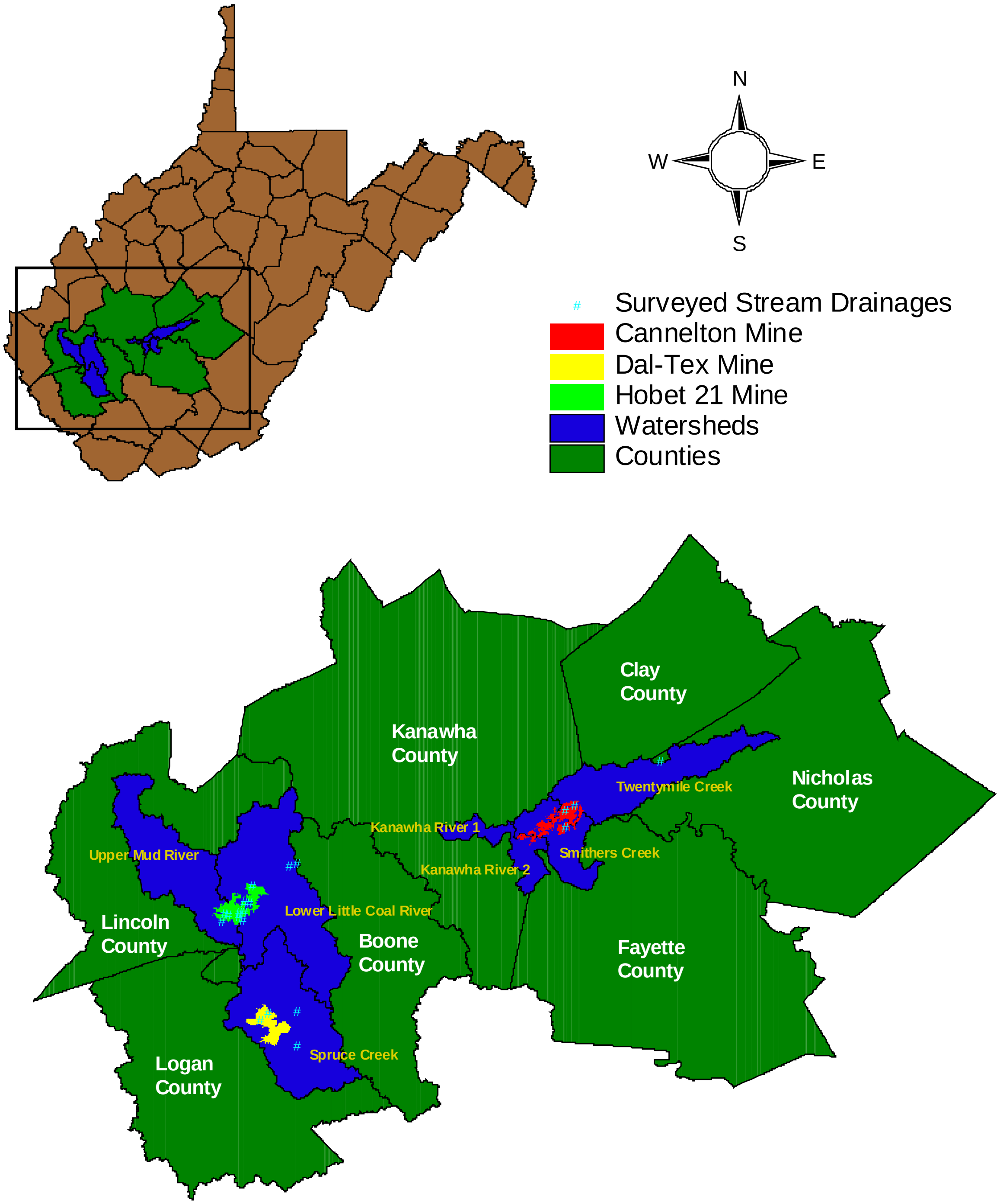

Figure 1. Location of mountaintop removal mine sites within watersheds in southern West Virginia.

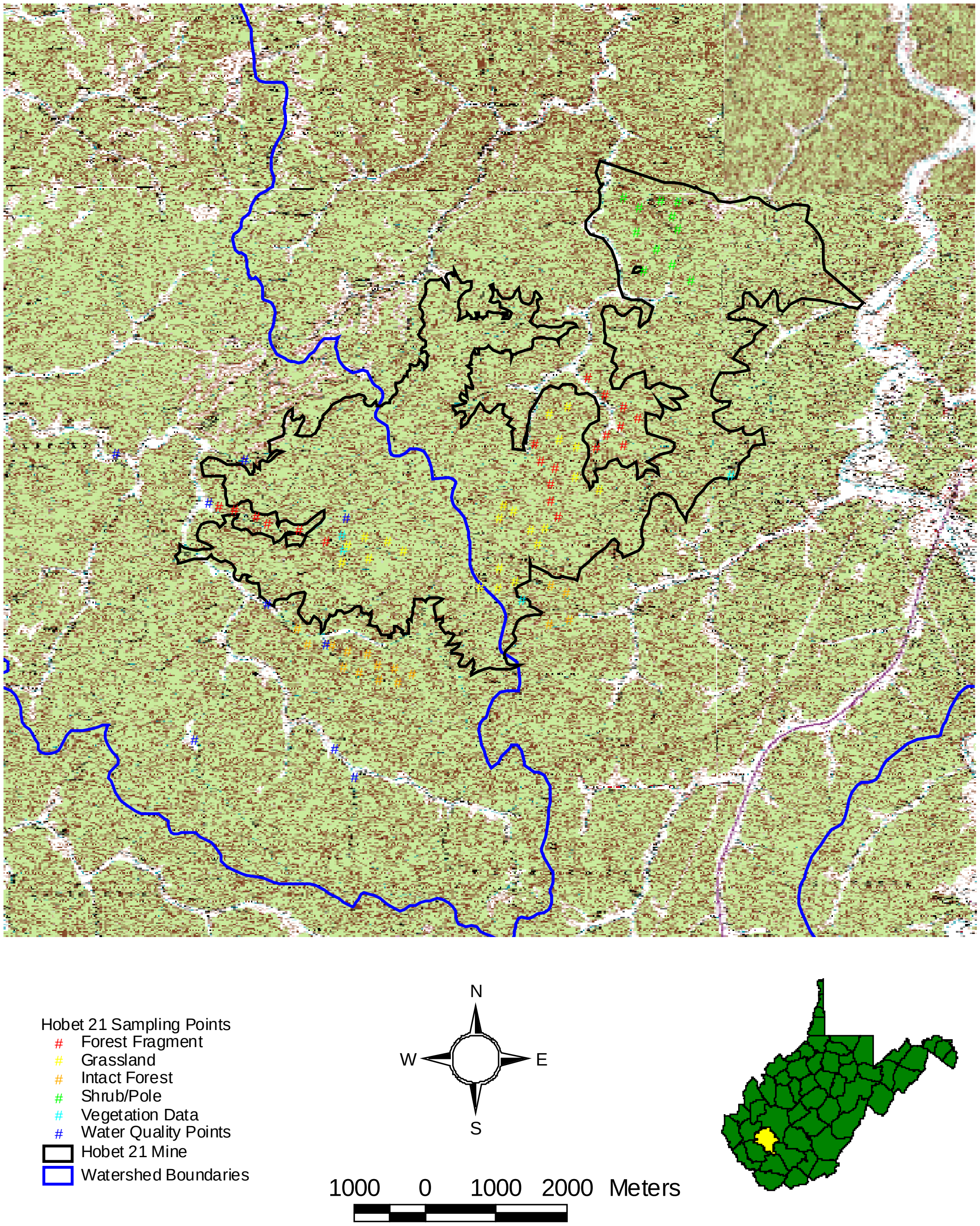

Figure 2. Topographic map of Hobet 21 mountaintop removal mine with locations of sampling points in Boone County, West Virginia.

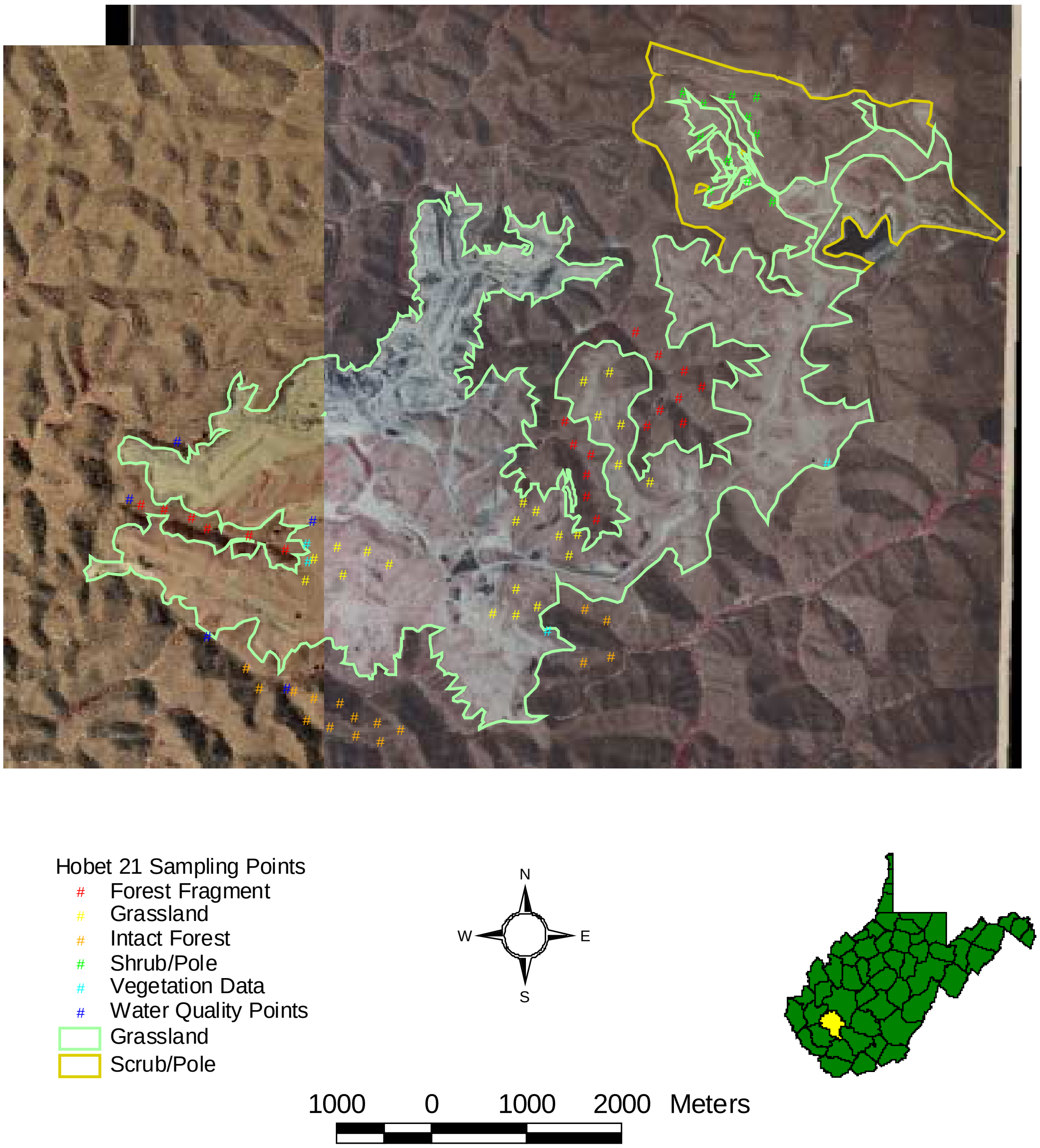

Figure 3. Aerial photograph of Hobet 21 mountaintop removal mine with locations of sampling points in Boone County, West Virginia.

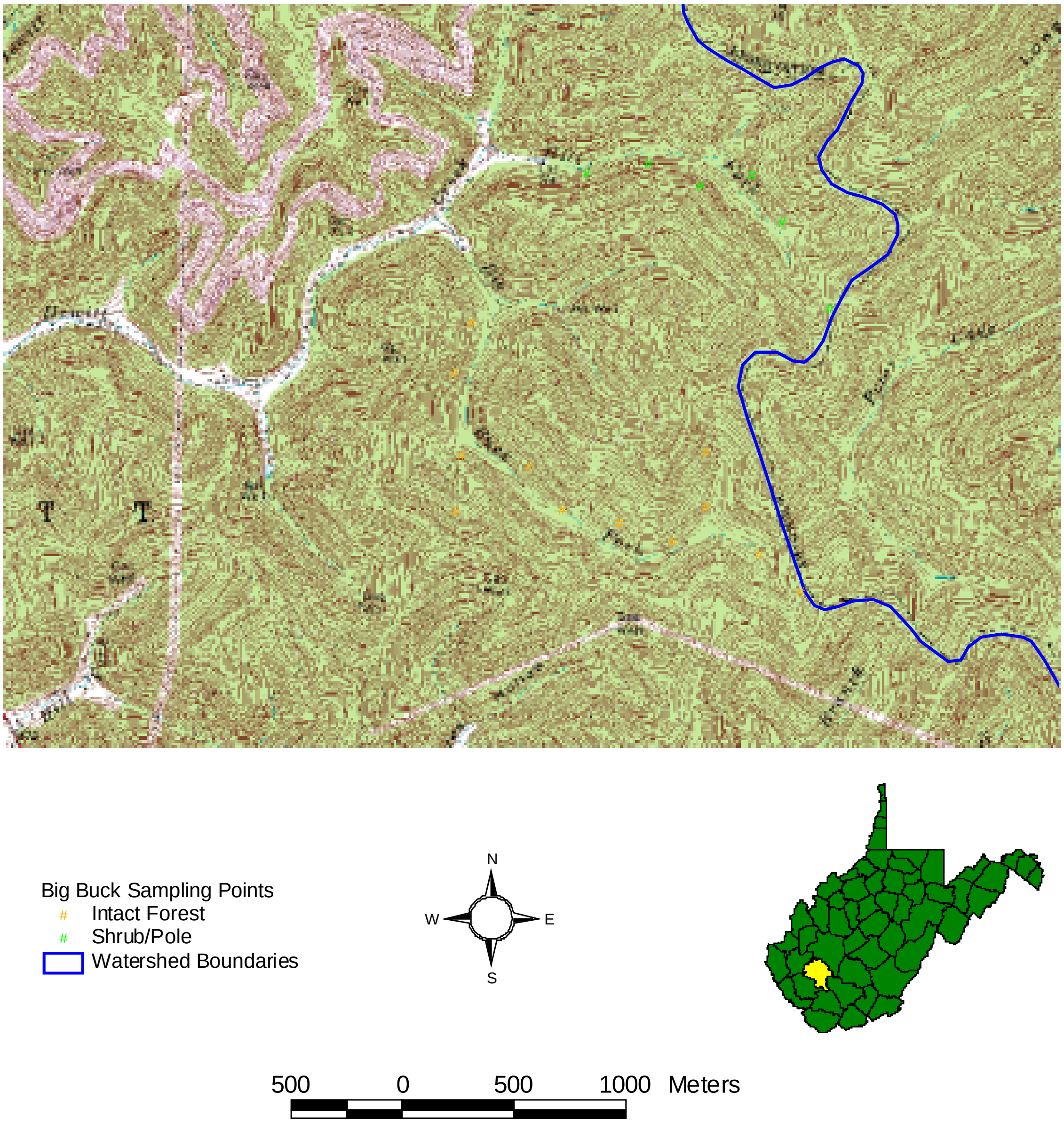

Figure 4. Topographic map of sampling points located along Big Buck Fork (intact forest) and Hill Fork drainages (shrub/pole) in Boone County, West Virginia.

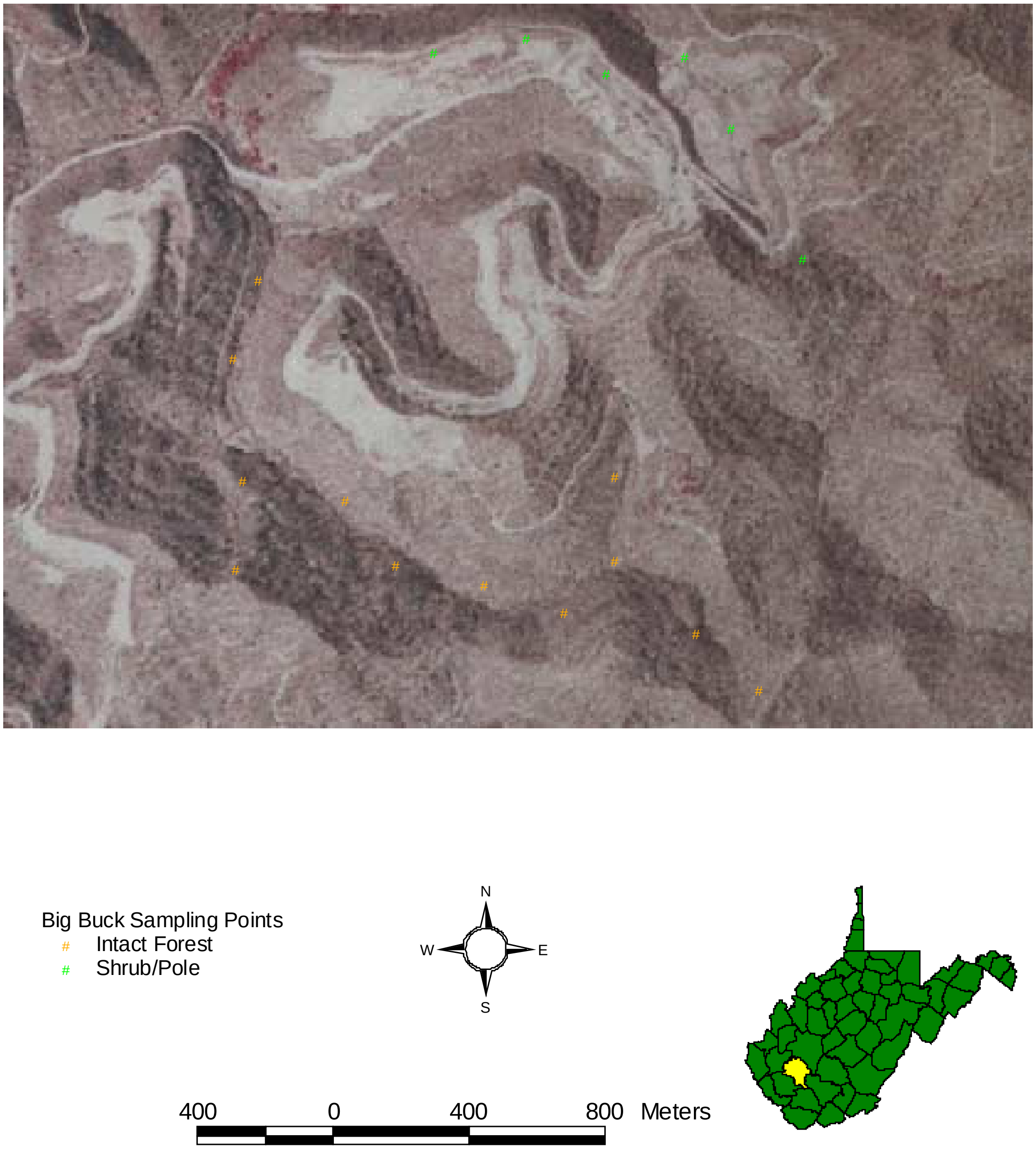

Figure 5. Aerial photograph of sampling points located along Big Buck Fork (intact forest) and Hill Fork drainages (shrub/pole) in Boone County, West Virginia.

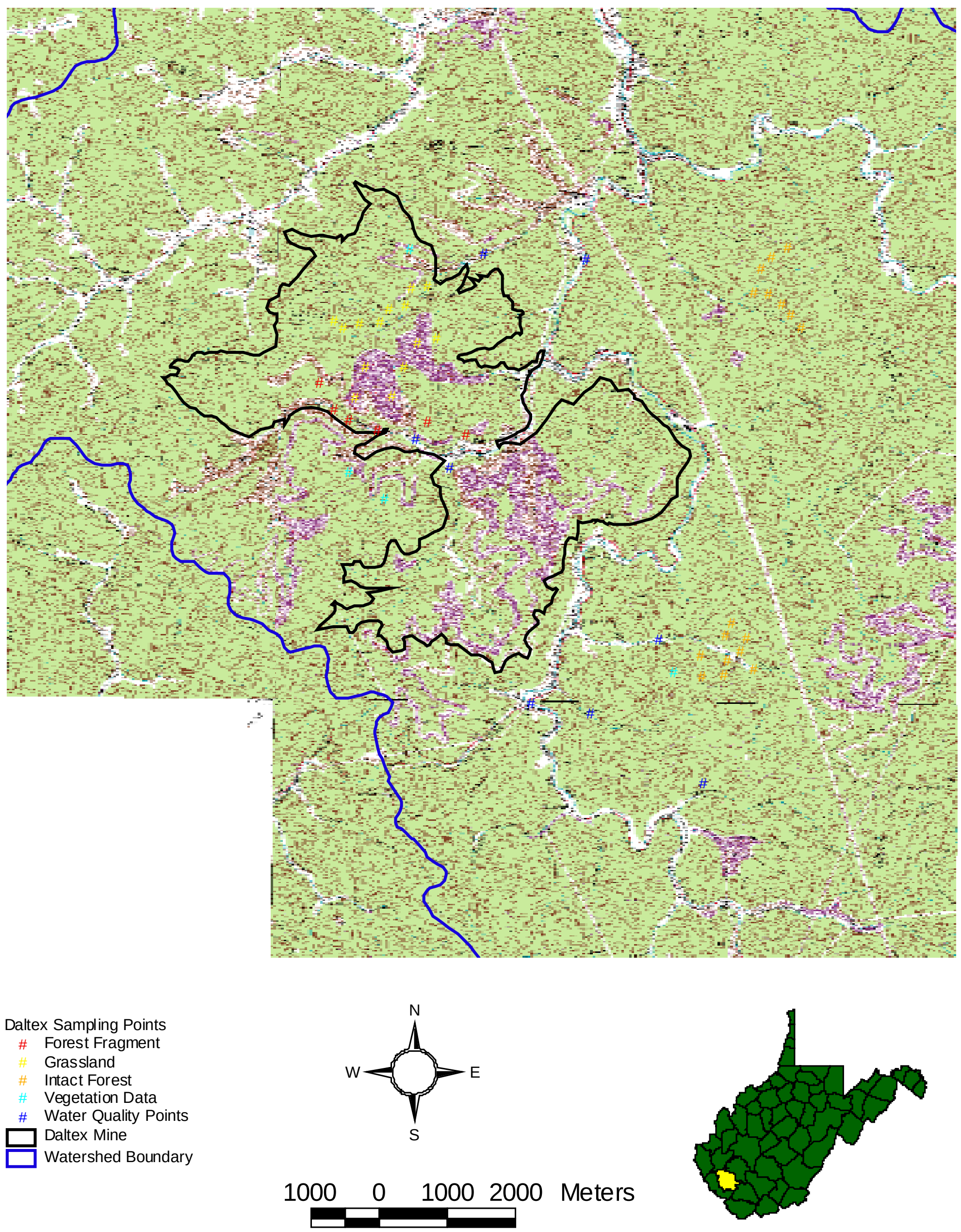

Figure 6. Topographic map of Daltex mountaintop removal mine with locations of sampling points in Logan County, West Virginia.

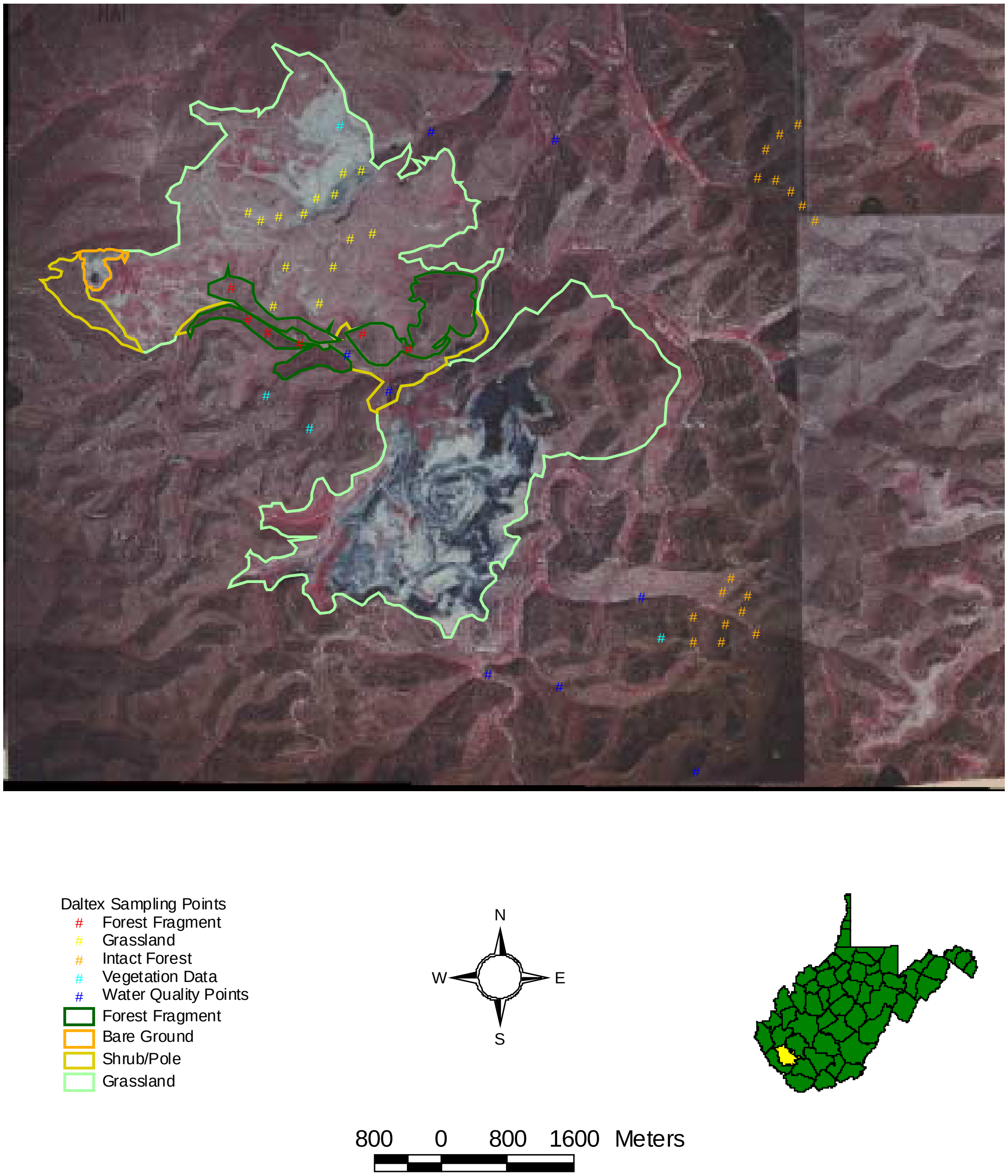

Figure 7. Aerial photograph of Daltex mountaintop removal mine with locations of sampling points in Logan County, West Virginia.

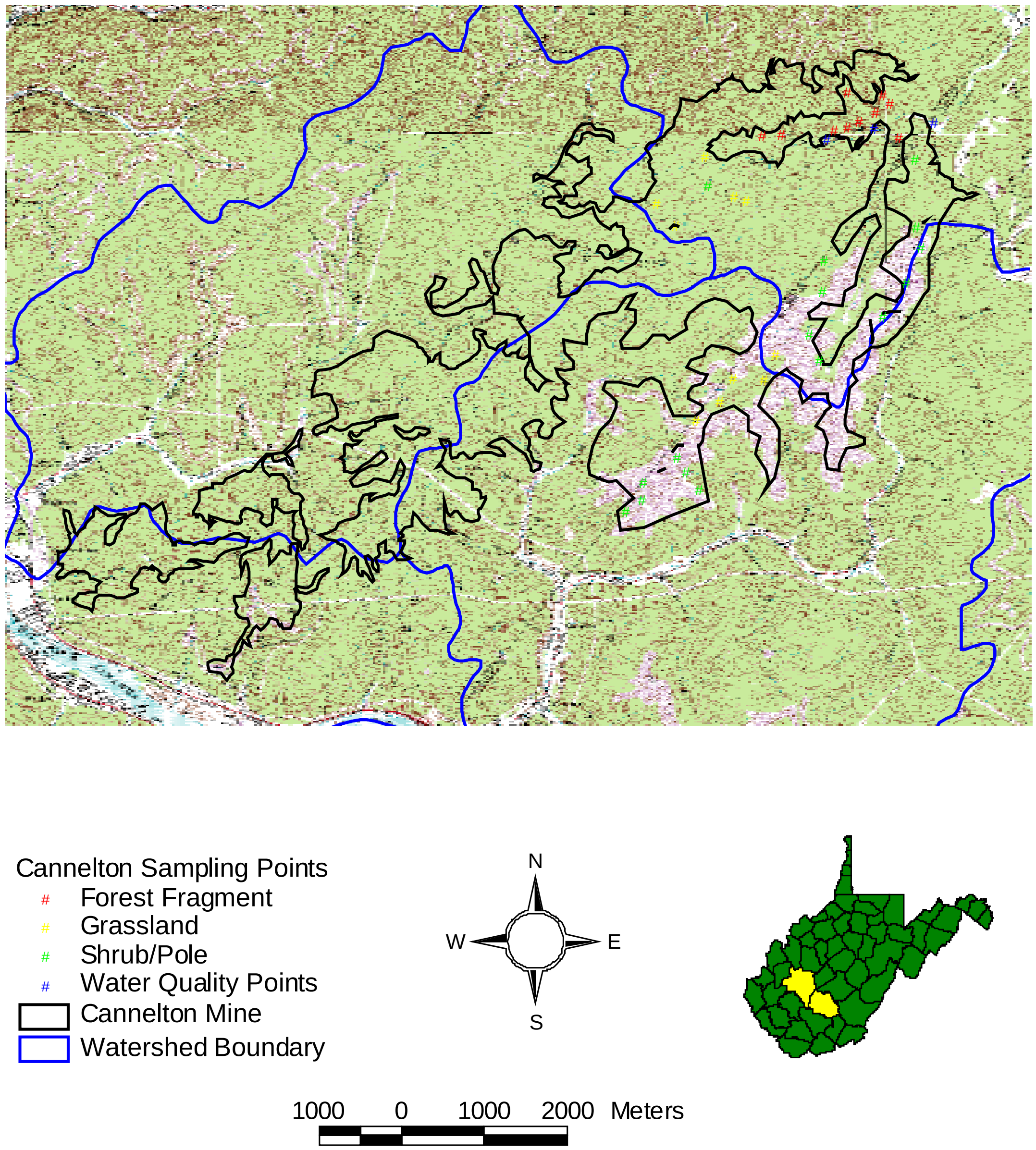

Figure 8. Topographic map of Cannelton mountaintop removal mine with locations of sampling points in Kanawha and Fayette Counties, West Virginia.

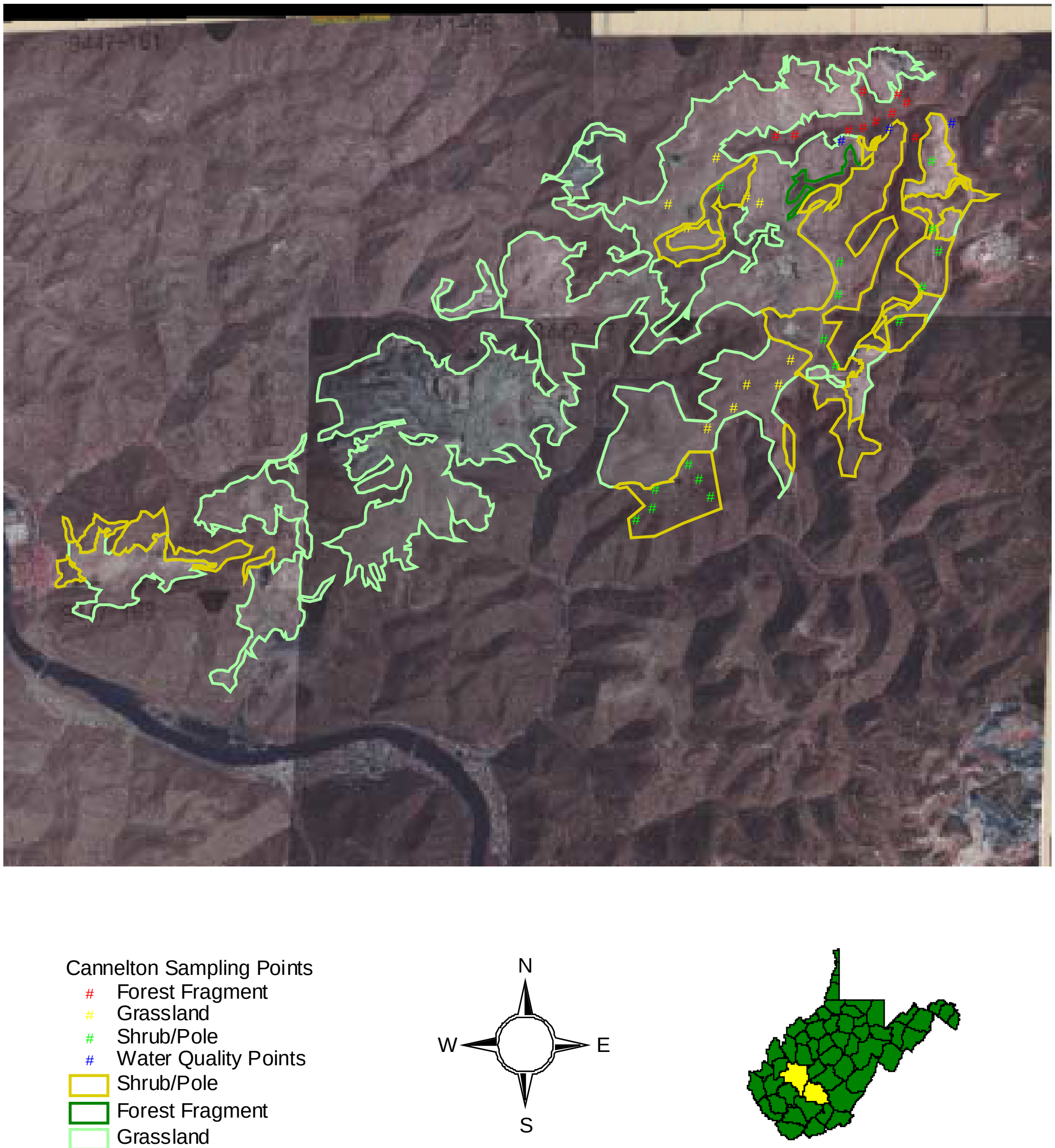

Figure 9. Aerial photograph of Cannelton mountaintop removal mine with locations of sampling points in Kanawha and Fayette Counties, West Virginia.

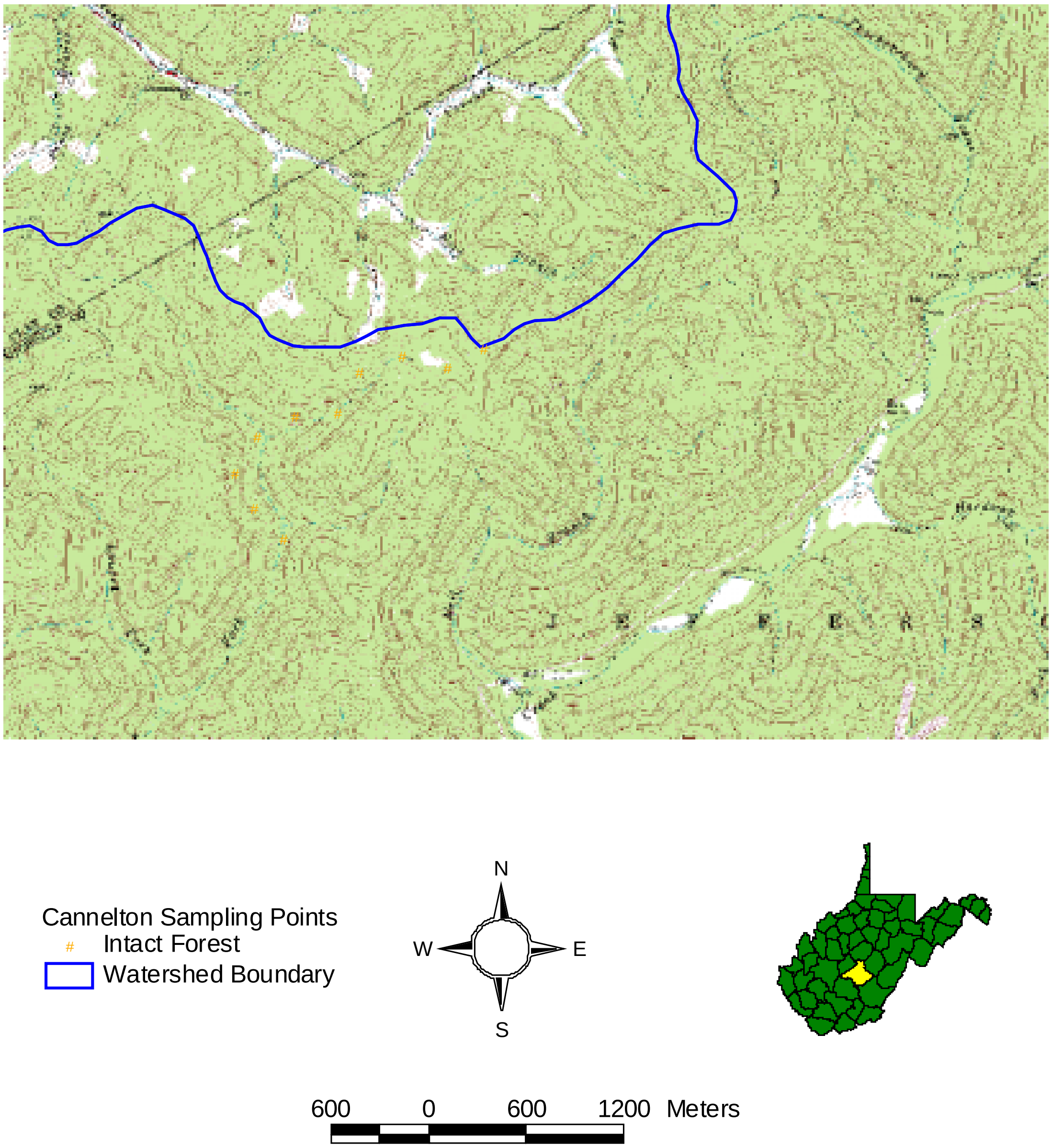

Figure 10. Topographic map of sampling points located along Ash Fork (intact forest) in Nicholas County, West Virginia.

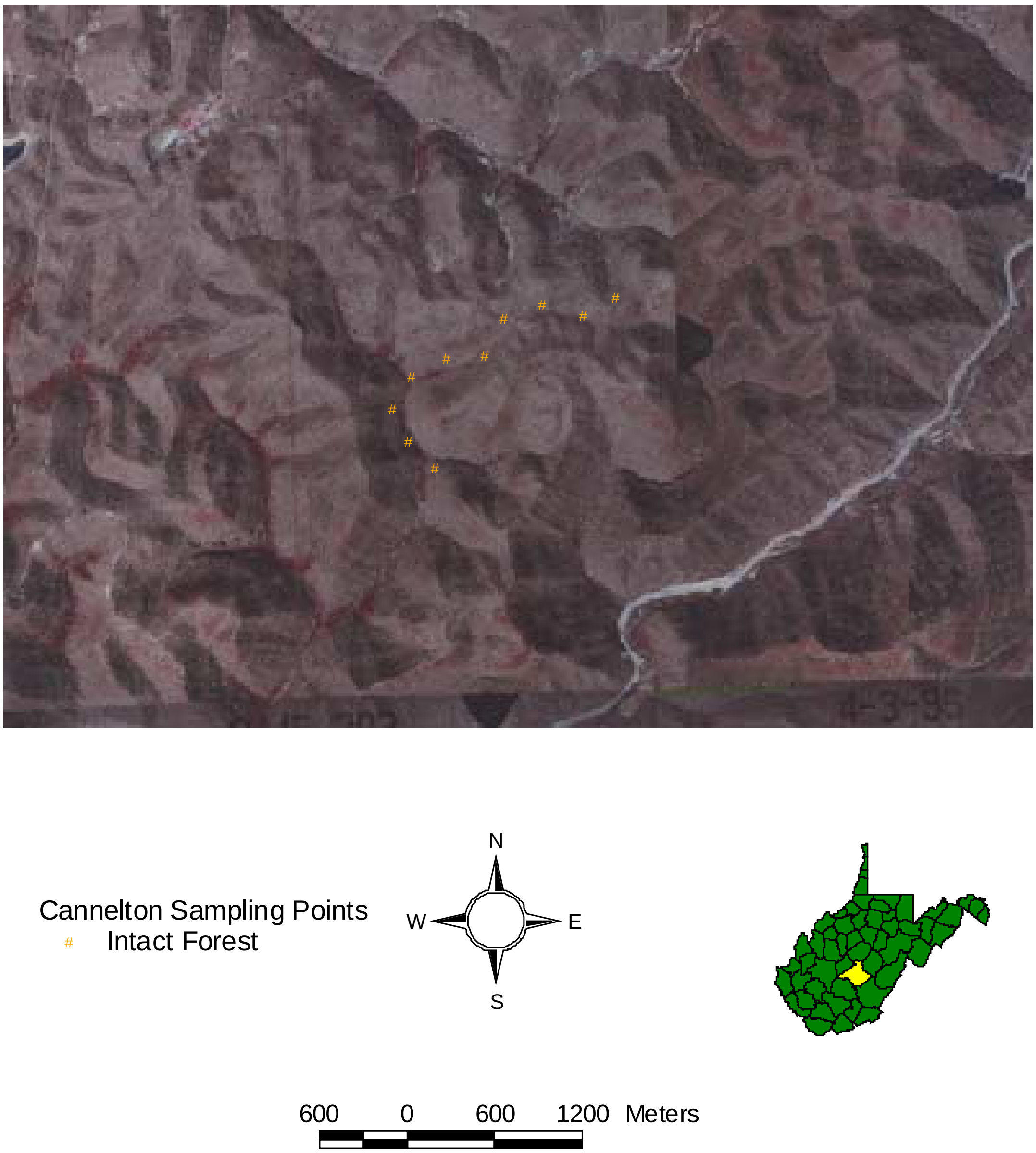

Figure 11. Aerial photograph of sampling points along Ash Fork (intact forest) in Nicholas County, West Virginia.

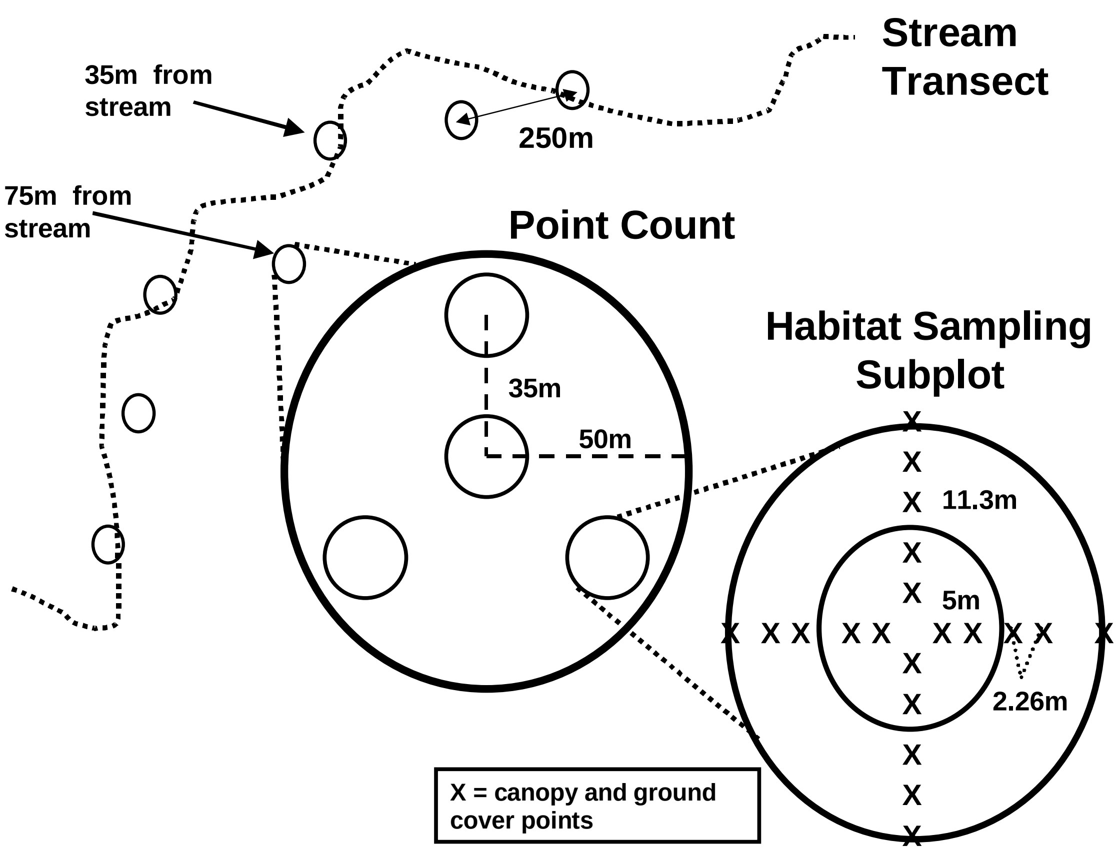

Figure 12. Placement of point count plots along streams and layout of vegetation sampling subplots within the 50-m radius point count plot.

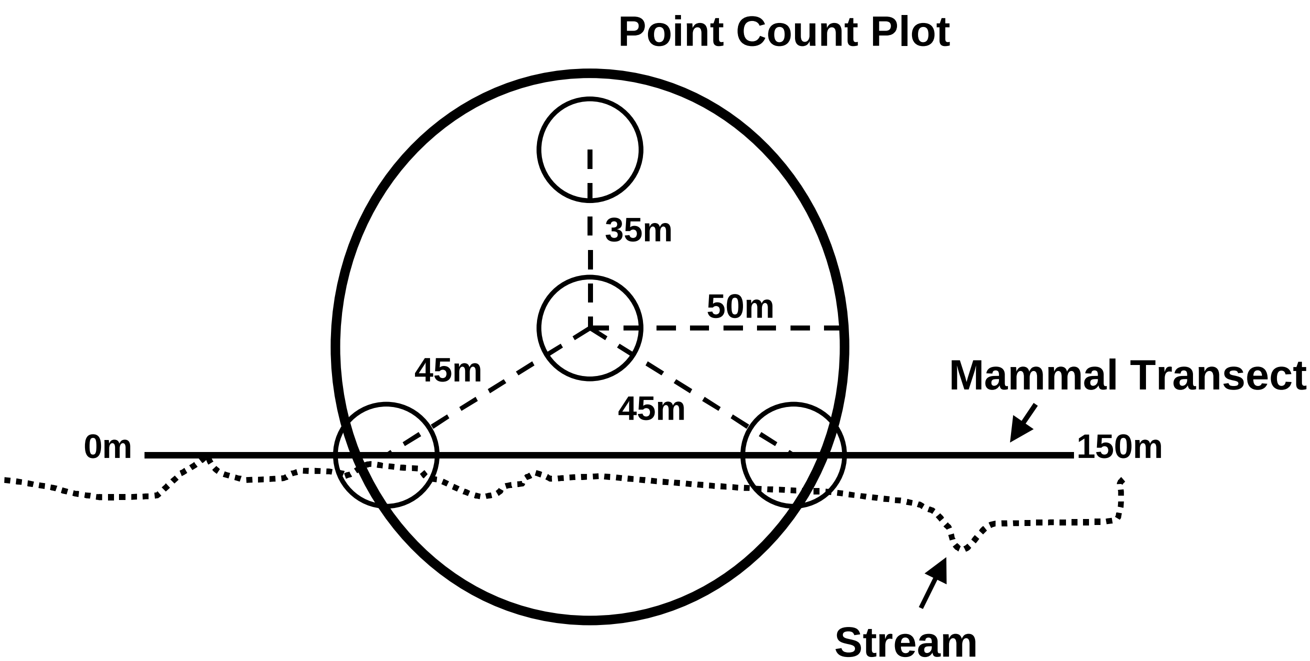

Figure 13. Layout of small mammal transects in relation to the bird point count plot and stream.

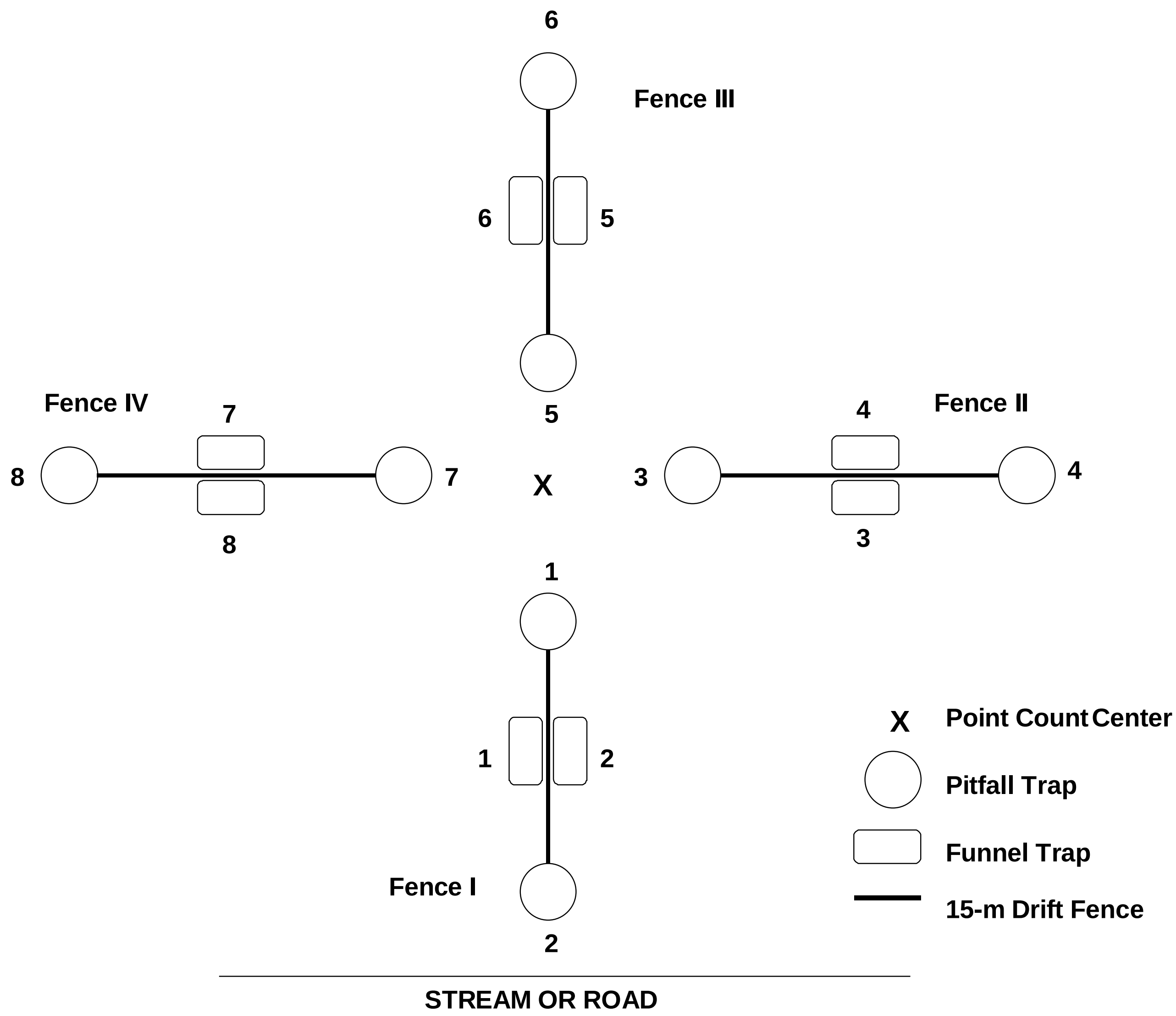

Figure 14. Placement of herpetofaunal drift fence array relative to songbird point count station.

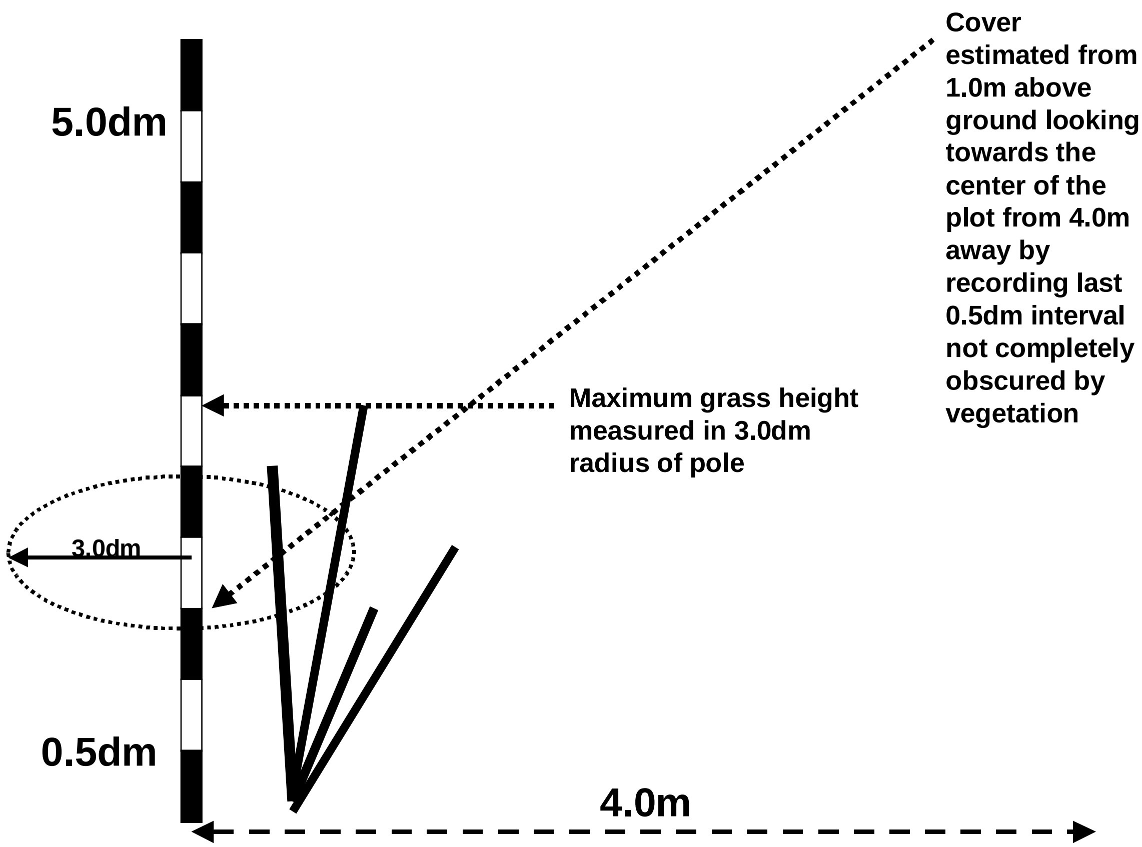

Figure 15. Example of how a Robel pole is used to measure vegetative cover and grass height.

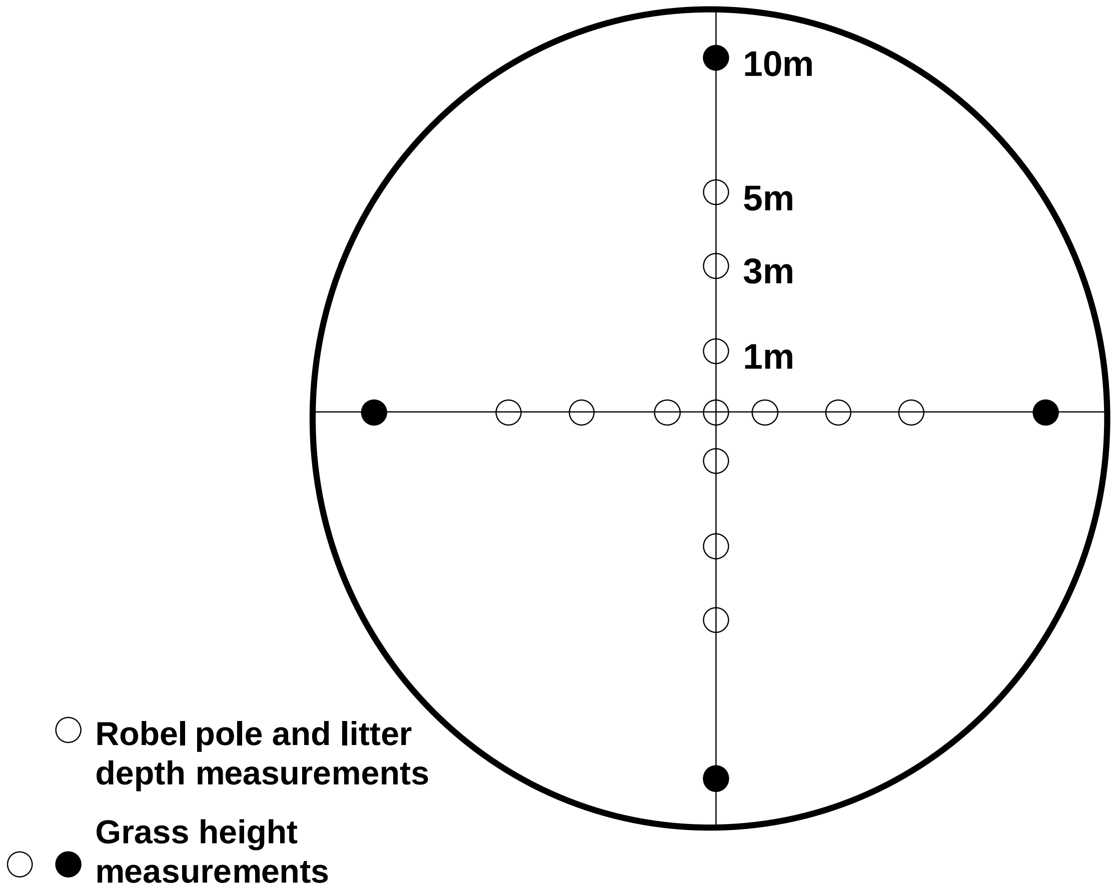

Figure 16. Sampling points on grassland vegetation subplot for vegetative cover and grass height measurements (Robel pole) and litter depth measurements.

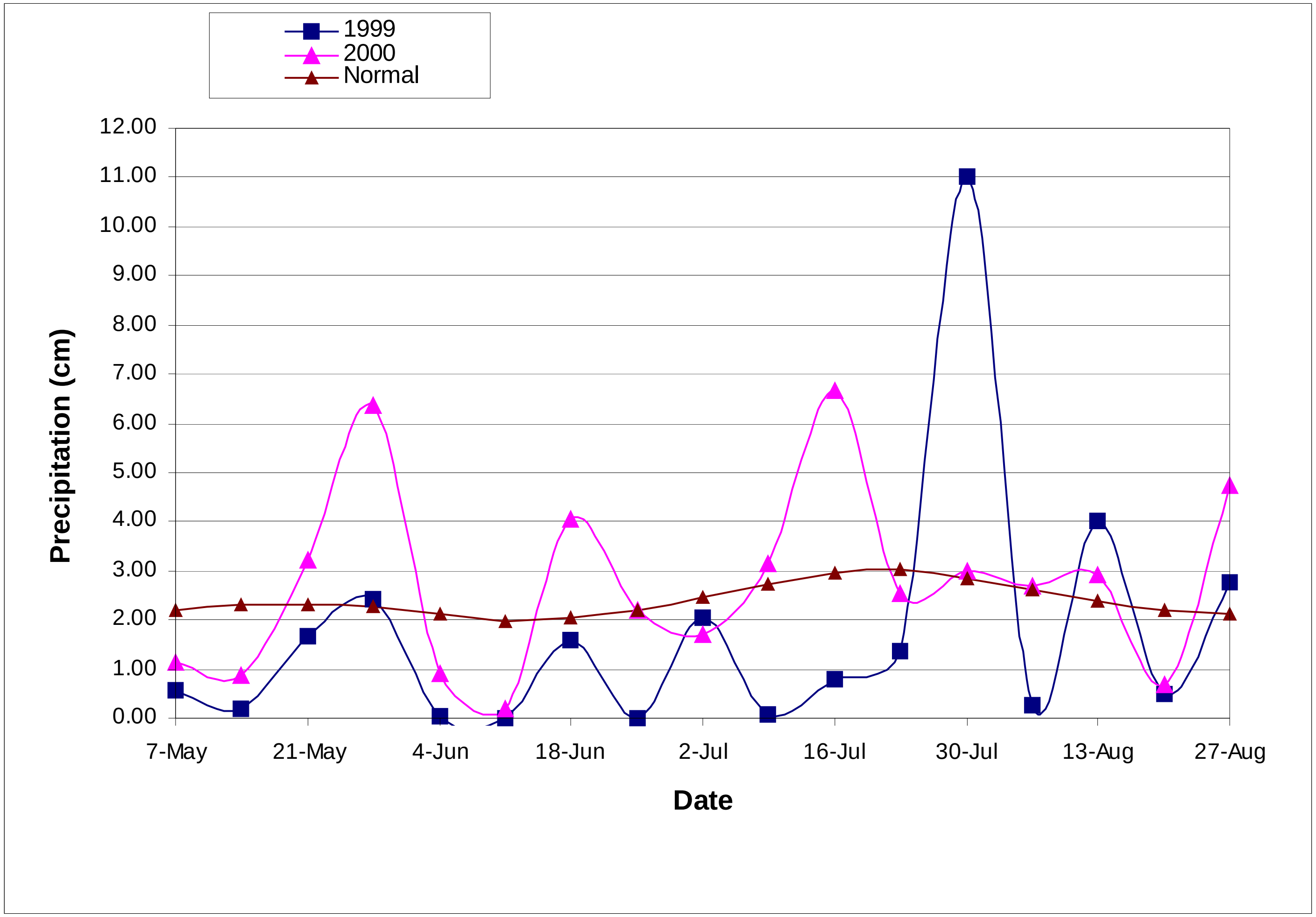

Fig. 17. Weekly precipitation reported in Charleston, West Virginia from May to August in 1999 and 2000. Total precipitation from May to August was 29.2 cm in 1999 and 47.0 cm in 2000.

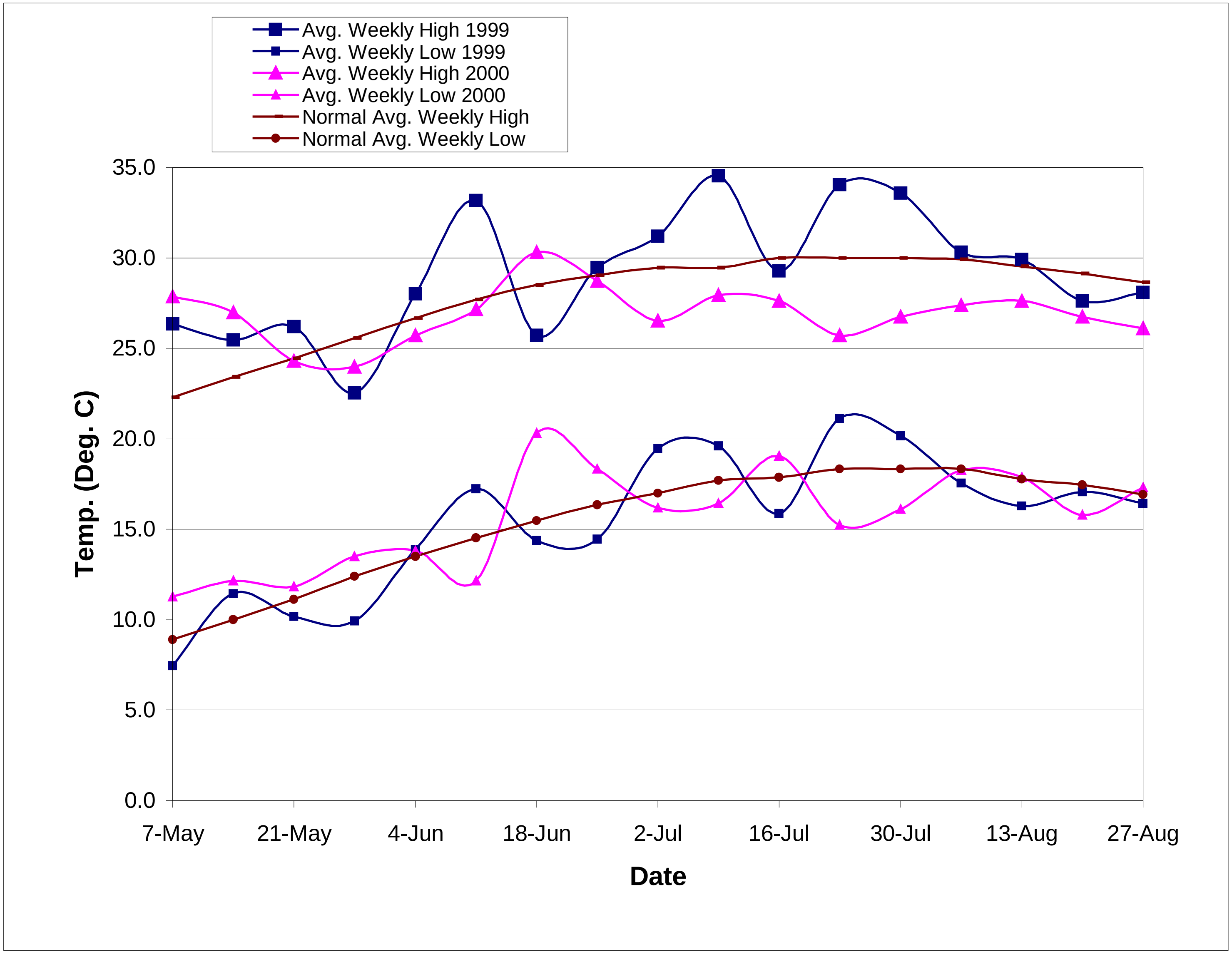

Fig. 18. Average weekly high and low temperatures recorded in Charleston, West Virginia from May to August 1999 and 2000. In 1999 the average high was 29.1 degrees C while the low was 15.4. In 2000, the average high was 26.9 degrees C and the low was 15.9.

Appendix 1. Orders, common names, and scientific names of all bird species mentioned in the text.

Order/Species	Scientific Name
<u>Order Podicepediformes</u>	
Pied-billed Grebe	*Podilymbus podiceps*
<u>Order Pelecaniformes</u>	
Double-crested Cormorant	*Phalacrocorax auritus*
<u>Order Ciconiiformes</u>	
American Bittern	*Botaurus lentiginosus*
Great Blue Heron	*Ardea herodias*
Great Egret	*Casmerodius albus*
Cattle Egret	*Bubulcus ibis*
Green-backed Heron	*Butorides striatus*
Yellow-crowned Night-Heron	*Nycticorax violaceus*
<u>Order Anseriformes</u>	
Mute Swan*	*Cygnus olor*
Canada Goose	*Branta canadensis*
Green-winged Teal	*Anas crecca*
American Black Duck	*Anas rubripes*
Mallard	*Anas platyrhynchos*
Northern Pintail	*Anas acuta*
Blue-winged Teal	*Anas discors*
Northern Shoveler	*Anas clypeata*
Gadwall	*Anas strepera*
American Wigeon	*Anas americana*
Redhead	*Aythya americana*
Ring-necked Duck	*Aythya collaris*
Lesser Scaup	*Aythya affinis*
Common Goldeneye	*Bucephala clangula*
Bufflehead	*Bucephala albeola*
Hooded Merganser	*Lophodytes cucullatus*
Common Merganser	*Mergus merganser*
<u>Order Falconiformes</u>	
Black Vulture	*Coragyps stratus*
Turkey Vulture	*Cathartes aura*

Order/Species	Scientific Name
Northern Harrier	*Circus cyaneus*
Sharp-shinned Hawk	*Accipiter striatus*
Cooper's Hawk	*Accipiter cooperii*
Northern Goshawk	*Accipiter gentilis*
Red-shouldered hawk	*Buteo lineatus*
Broad-winged Hawk	*Buteo platypterus*
Red-tailed Hawk	*Buteo jamaicensis*
Rough-legged Hawk	*Buteo lagopus*
American Kestrel	*Falco sparverius*
Peregrine Falcon	*Falco peregrinus*
<u>Order Galliformes</u>	
Ring-necked Pheasant*	*Phasianus colchicus*
Ruffed Grouse	*Bonasa umbellus*
Wild Turkey	*Meleagris gallopavo*
Northern Bobwhite	*Colinus virginianus*
<u>Order Gruiformes</u>	
King Rail	*Rallus elegans*
Sora	*Porzana carolina*
Common Moorhen	*Gallinula chloropus*
American Coot	*Fulica americana*
<u>Order Charadriiformes</u>	
American Golden-plover	*Pluvialis dominica*
Killdeer	*Charadrius vociferous*
Greater Yellowlegs	*Tringa melanoleuca*
Lesser Yellowlegs	*Tringa flavipes*
Solitary Sandpiper	*Tringa solitaria*
Spotted Sanpiper	*Actitis macularia*
Semipalmated Sandpiper	*Calidris pusilla*
Western Sandpiper	*Calidris mauri*
Least Sandpiper	*Calidris minutilla*
White-rumped Sandpiper	*Calidris fuscicollis*
Baird's Sandpiper	*Calidris bairdii*
Pectoral Sandpiper	*Calidris melanotos*

Appendix 1. Continued.

Order/Species	Scientific Name
Common Snipe	*Gallinago gallinago*
American Woodcock	*Scolopax minor*
Order Columbiformes	
Rock Dove*	*Columba livia*
Mourning Dove	*Zenaida macroura*
Order Cuculiformes	
Black-billed Cuckoo	*Coccyzus erthropthalmus*
Yellow-billed Cuckoo	*Coccyzus americanus*
Order Strigiformes	
Eastern Screech-Owl	*Otus asio*
Great Horned Owl	*Bulbo virginianus*
Barred Owl	*Strix varia*
Short-eared Owl	*Asio flammeus*
Order Caprimulgiformes	
Common Nighthawk	*Chordeiles minor*
Whip-poor-will	*Caprimulgus vociferus*
Order Apodiformes	
Chimney Swift	*Chaetura pelagica*
Ruby-throated Hummingbird	*Archilocus colubris*
Order Coraciiformes	
Belted Kingfisher	*Ceryle torquata*
Order Piciformes	
Red-headed Woodpecker	*Melanerpes erythrocephalus*
Red-bellied Woodpecker	*Melanerpes carolinus*
Downy Woodpecker	*Picoides pubescens*
Hairy Woodpecker	*Picoides villosus*
Northern Flicker	*Colaptes auratus*
Pileated Woodpecker	*Dryocopus pileatus*

Order/Species	Scientific Name
Order Passeriformes	
Acadian Flycatcher	*Empidonax virescens*
Willow Flycatcher	*Empidonax traillii*
Least Flycatcher	*Empidonax minimus*
Eastern Phoebe	*Sayornis phoebe*
Great Crested Flycatcher	*Myiarchus crinitus*
Eastern Kingbird	*Tyrannus tyrannus*
Horned Lark	*Eremophila alpestris*
Purple Martin	*Progne subis*
Tree Swallow	*Tachycineta bicolor*
Northern Rough-winged Swallow	*Stelgidopteryx serripennis*
Bank Swallow	*Riparia riparia*
Cliff Swallow	*Petrochelidon pyrrhonota*
Barn Swallow	*Hirundo rustica*
Blue Jay	*Cyanocitta cristata*
American Crow	*Corvus brachyrhynchos*
Common Raven	*Corvus corax*
Black-capped Chickadee	*Poecile atricapilla*
Carolina Chickadee	*Poecile carolinensis*
Tufted Titmouse	*Baeolophus bicolor*
White-breasted Nuthatch	*Sitta carolinensis*
Brown Creeper	*Certhia americana*
Carolina Wren	*Thryothorus ludovicianus*
House Wren	*Troglodytes aedon*
Winter Wren	*Troglodytes troglodytes*
Blue-gray Gnatcatcher	*Polioptila caerulea*
Eastern Bluebird	*Sialia sialis*
Veery	*Catharus fuscescens*
Wood Thrush	*Hylocichla mustelina*
American Robin	*Turdus migratorius*
Gray Catbird	*Dumatella carolinensis*
Northern Mockingbird	*Mimus polyglottos*
Brown Thrasher	*Toxostoma rufum*
European Starling*	*Sturnus vulgaris*
White-eyed Vireo	*Vireo griseus*
Blue-headed Vireo	*Vireo solitarius*

Appendix 1. Continued.

Order/Species	Scientific Name
Warbling Vireo	*Vireo gilvus*
Yellow-throated Vireo	*Vireo flavifrons*
Eastern Wood-Pewee	*Contopus virens*
Red-eyed Vireo	*Vireo olivaceus*
Blue-winged Warbler	*Vermivora pinus*
Golden-winged Warbler	*Vermivora chrysoptera*
Northern Parula	*Parula americana*
Yellow Warbler	*Dendroica petechia*
Chestnut-sided Warbler	*Dendroica pensylvanica*
Black-throated Blue Warbler	*Dendroica caerulescens*
Black-throated Green Warbler	*Dendroica virens*
Yellow-throated Warbler	*Dendroica dominica*
Pine Warbler	*Dendroica pinus*
Prairie Warbler	*Dendroica discolor*
Palm Warbler	*Dendroica palmarum*
Cerulean Warbler	*Dendroica cerulea*
Black-and-white Warbler	*Mniotilta varia*
American Redstart	*Setophaga ruticilla*
Worm-eating Warbler	*Helmitheros vermivorus*
Swainson's Warbler	*Limnothlypis swainsonii*
Ovenbird	*Seiurus aurocapillus*
Louisiana Waterthrush	*Seiurus motacilla*
Kentucky Warbler	*Oporornis formosus*
Common Yellowthroat	*Geothlypis trichas*
Hooded Warbler	*Wilsonia citrina*
Canada Warbler	*Wilsonia canadensis*
Yellow-breasted Chat	*Icteria virens*
Summer Tanager	*Piranga rubra*
Scarlet Tanager	*Piranga olivacea*
Northern Cardinal	*Cardinalis cardinalis*
Rose-breasted Grosbeak	*Pheucticus ludovicianus*
Blue Grosbeak	*Guiraca caerulea*
Indigo Bunting	*Passerina cyanea*
Dickcissel	*Spiza americana*
Eastern Towhee	*Pipilo erythropthalmus*
Chipping Sparrow	*Spizella passerina*
Field Sparrow	*Spizella pusilla*
Dark-eyed Junco	*Junco hyemalis*
Bobolink	*Dolichonyx oryzivorus*
Red-winged Blackbird	*Agelaius phoeniceus*
Eastern Meadowlark	*Sturnella magna*
Common Grackle	*Quiscalus quiscula*
Cedar Waxwing	*Bombycilla cedrorum*
Brown-headed Cowbird	*Molothrus ater*
Orchard Oriole	*Icterus spurius*
Baltimore Oriole	*Icterus galbula*
Purple Finch	*Carpodacus purpureus*
House Finch*	*Carpodacus mexicanus*
American Goldfinch	*Carduelis tristis*
House Sparrow*	*Passer domesticus*

Appendix 2. Common and scientific names of woody plants found on sampling points in grassland, shrub/pole, fragmented forest, and intact forest treatments.

		Treatment										
		Grassland			Shrub/pole		Fragmented Forest			Intact Forest		
Common Name	Scientific Name[a]	Can.	Dal.	Hob.	Can.	Hob.	Can.	Dal.	Hob.	Can.	Dal.	Hob.
American basswood	*Tilia americana*						x	x	x	x	x	x
American beech	*Fagus grandifolia*					x[b]	x	x	x	x	x	x
American chestnut	*Castanea dentata*						x		x	x	x	x
Common elderberry	*Sambucus canadensis*			x				x				
American elm	*Ulmus americana*							x			x	x
American hazelnut	*Corlyus americana*							x	x		x	x
American sycamore	*Platanus occidentalis*				x	x		x	x		x	
Autumn olive	*Elaegnus umbellata*	x	x	x	x			x				
Bicolor lespedeza	*Lespedeza bicolor*	x	x	x		x						
Bitternut hickory	*Carya cordiformis*						x	x	x		x	x
Blackberry/raspberry	*Rubus spp.*	x	x	x	x	x	x	x	x		x	x
Black birch	*Betula lenta*				x	x	x	x	x	x	x	x
Black cherry	*Prunus serotina*				x	x[b]	x		x		x	x
Black gum	*Nyssa sylvatica*				x		x	x	x	x	x	x
Black locust	*Robinia pseudoacacia*		x	x	x	x	x	x	x	x	x	x
Black oak	*Quercus velutina*					x	x	x	x	x	x	x
Blueberry	*Vaccinium spp.*				x		x	x	x			x
Black walnut	*Juglans nigra*							x	x		x	x
Box elder	*Acer negundo*					x						
Buffalo nut	*Pyrularia pubera*						x		x		x	
Chestnut oak	*Quercus prinus*							x	x	x	x	x
Cucumber magnolia	*Magnolia acuminata*					x[b]	x	x	x	x	x	x
Eastern hemlock	*Tsuga canadensis*						x		x	x	x	x
Eastern redbud	*Cercis canadensis*					x[b]	x	x	x		x	x
Eastern red cedar	*Juniperus virginiana*			x								
European black alder	*Alnus glutinosa*			x	x	x						
Flame Azalea	*Rhododendro calendulaceum*						x		x		x	x
Flowering dogwood	*Cornus florida*					x	x	x	x	x	x	x
Green ash	*Fraxinus pennsylvanica*				x	x	x	x	x	x	x	x
Greenbrier	*Smilax spp.*					x	x	x	x	x	x	x

Appendix 2. Continued.

		Treatment										
		Grassland			Shrub/pole		Fragmented Forest			Intact Forest		
Common Name	Scientific Name[a]	Can.	Dal.	Hob.	Can.	Hob.	Can.	Dal.	Hob.	Can.	Dal.	Hob.
Gray dogwood	*Cornus racemosa*							x				x
Hawthorn species	*Crataegus spp.*								x			
Hercule’s club	*Aralia spinosa*										x	
Honeysuckle	*Lonicera spp.*								x			
Ironwood	*Carpinus caroliniana*						x	x	x	x	x	x
Loblolly pine	*Pinus taeda*								x			
Multiflora rose	*Rosa multiflora*		x	x	x	x	x	x	x	x		x
Maple leaf viburnum	*Viburnum acerifolium*						x		x		x	x
Mockernut hickory	*Carya tomentosa*											x
Mountain laurel	*Kalmia latifolia*						x		x	x		x
Musclewood	*Ostyra virginiana*						x	x			x	x
Northern red oak	*Quercus rubra*						x	x	x	x	x	x
Ohio buckeye	*Aesculus glabra*							x				
Persimmon	*Diospyros virginiana*						x	x	x			
Pawpaw	*Asimina triloba*							x	x	x	x	x
Pignut hickory	*Carya glabra*						x	x	x	x	x	x
Pitch pine	*Pinus rigida*				x							
Poison ivy	*Toxicodendron radicans*				x	x	x	x		x	x	x
Princess tree	*Paulownia tomentosa*		x		x							
Red maple	*Acer rubrum*			x	x	x	x	x		x	x	x
Red mulberry	*Morus rubra*										x	
Red pine	*Pinus resinosa*				x							
River birch	*Betula nigra*							x	x			
Rhododendron	*Rhododendron maximum*						x					
Sassafras	*Sassafras albidum*						x	x	x	x	x	x
Scarlet Oak	*Quercus coccinea*						x	x	x		x	x
Scotch pine	*Pinus sylvestris*			x	x							
Serviceberry	*Amelanchier spp.*				x		x	x	x		x	x
Shagbark hickory	*Carya ovata*						x	x	x	x	x	x
Slippery elm	*Ulmus rubra*						x	x	x	x	x	x
Smooth Sumac	*Rhus glabra*				x							
Spicebush	*Lindera benzoin*						x			x	x	x

Appendix 2. Continued.

		Treatment										
		Grassland			Shrub/pole		Fragmented Forest			Intact Forest		
Common Name	Scientific Name[a]	Can.	Dal.	Hob.	Can.	Hob.	Can.	Dal.	Hob.	Can.	Dal.	Hob.
Sourwood	*Oxydendrum arboreum*			x	x	x	x	x	x	x	x	x
Staghorn sumac	*Rhus typhina*					x[b]					x	
Sugar maple	*Acer saccharum*				x	x	x	x	x	x	x	x
Sweetgum	*Liquidambar styraciflua*				x					x	x	x
Tree of heaven	*Ailanthus altissima*				x	x	x	x	x		x	x
Tuliptree	*Liriodendron tulipifera*				x	x[b]	x	x	x	x	x	x
Umbrella magnolia	*Magnolia tripetala*						x	x	x	x	x	x
Virginia Creeper	*Parthenocissus quinquefolia*						x	x	x	x	x	x
Virginia pine	*Pinus virginiana*				x				x			
White ash	*Fraxinus americana*			x		x[b]	x	x	x	x	x	x
White oak	*Quercus alba*						x	x	x	x	x	x
White pine	*Pinus strobus*			x	x	x						
Wild grape	*Vitis spp.*				x					x	x	x
Willow species	*Salix spp.*			x								
Witchhazel	*Hamamelis virginiana*						x	x	x	x	x	x
Wild hydrangea	*Hydrangea arborescens*						x	x	x	x	x	x
Wild rose	*Rosa spp.*										x	
Winged sumac	*Rhus copallina*					x	x		x		x	
Yellow birch	*Betula allegheniensis*							x	x	x	x	x

[a] Nomenclature follows Strausbaugh and Core (1977).
[b] Species only found in the Mud River/Coal River watersheds at the Hill Fork site (a valleyfill associated with a contour mine).

Appendix 3. Mean abundance of songbird species and guilds in grassland, shrub/pole, fragmented forest, and intact forest treatments on the Hobet and Daltex mine sites in 1999.

	Treatment						
	Grasslands		Shrub/pole	Fragmented Forest		Intact Forest	
Species/Guild	Hobet	Daltex	Hobet	Hobet	Daltex	Hobet	Daltex
Forest Interior Species							
Acadian Flycatcher	0.00	0.00	0.17	1.05	0.50	1.00	1.50
Black-throated Green Warbler	0.00	0.00	0.00	0.00	0.00	0.07	0.00
Blue-headed Vireo	0.00	0.00	0.00	0.20	0.50	0.39	0.63
Cerulean Warbler	0.00	0.00	0.00	0.20	0.25	0.39	0.25
Eastern Wood-pewee	0.00	0.00	0.00	0.00	0.00	0.04	0.00
Great Crested Flycatcher	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Kentucky Warbler	0.00	0.00	0.17	0.30	0.25	0.18	0.63
Louisiana Waterthrush	0.00	0.00	0.00	0.10	0.00	0.18	0.13
Ovenbird	0.00	0.00	0.00	0.65	0.00	0.93	1.25
Pileated Woodpecker	0.00	0.00	0.00	0.20	0.00	0.00	0.00
Scarlet Tanager	0.00	0.00	0.00	0.25	0.00	0.04	0.38
Summer Tanager	0.00	0.00	0.00	0.10	0.25	0.11	0.13
Swainson's Warbler	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Wood Thrush	0.00	0.00	0.00	0.85	0.50	0.43	0.50
Worm-eating Warbler	0.00	0.00	0.00	0.10	0.00	0.18	0.25
Yellow-throated Warbler	0.00	0.00	0.00	0.05	0.00	0.11	0.00
Interior-edge Species							
American Redstart	0.00	0.00	0.50	0.25	0.25	0.46	0.75
American Robin	0.00	0.00	0.00	0.05	0.00	0.00	0.00
Black-and-white Warbler	0.00	0.00	0.00	0.30	0.25	0.29	0.00
Black-capped Chickadee	0.00	0.00	0.00	0.05	0.00	0.04	0.00
Blue-gray Gnatcatcher	0.00	0.00	0.00	0.05	0.00	0.04	0.00
Carolina Chickadee	0.00	0.00	0.00	0.40	0.50	0.43	0.38
Carolina Wren	0.00	0.00	0.17	0.35	0.50	0.36	0.75
Dark-eyed Junco	0.00	0.00	0.00	0.05	0.00	0.00	0.00
Downy Woodpecker	0.00	0.00	0.00	0.05	0.25	0.04	0.13
Eastern Phoebe	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eastern Towhee	0.05	0.00	0.50	0.00	0.00	0.00	0.00
Hairy Woodpecker	0.00	0.00	0.00	0.00	0.00	0.14	0.00

Appendix 3. Continued.

	Treatment						
	Grasslands		Shrub/pole	Fragmented Forest		Intact Forest	
Species/Guild	Hobet	Daltex	Hobet	Hobet	Daltex	Hobet	Daltex
Hooded Warbler	0.00	0.00	0.33	0.20	0.00	0.29	0.88
Northern Flicker	0.00	0.00	0.00	0.10	0.00	0.07	0.00
Northern Parula	0.00	0.00	0.00	0.20	0.00	0.14	0.13
Red-bellied Woodpecker	0.00	0.00	0.00	0.00	0.25	0.11	0.00
Red-eyed Vireo	0.00	0.00	0.50	1.00	1.00	0.93	0.88
Ruby-throated Hummingbird	0.00	0.00	0.00	0.10	0.00	0.14	0.00
Tufted Titmouse	0.00	0.00	0.00	0.10	0.25	0.07	0.50
White-breasted Nuthatch	0.00	0.00	0.00	0.05	0.25	0.21	0.25
Yellow-billed Cuckoo	0.00	0.00	0.33	0.05	0.00	0.11	0.00
Yellow-throated Vireo	0.00	0.00	0.00	0.05	0.50	0.11	0.00
Edge Species							
American Crow	0.00	0.00	0.00	0.05	0.50	0.00	0.00
American Goldfinch	0.45	0.13	2.67	0.05	0.25	0.00	0.00
Baltimore Oriole	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Blue Grosbeak	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Blue Jay	0.05	0.00	0.00	0.10	0.00	0.04	0.00
Blue-winged Warbler	0.14	0.00	1.17	0.05	0.00	0.07	0.00
Brown Thrasher	0.09	0.13	0.17	0.00	0.00	0.00	0.00
Brown-headed Cowbird	0.00	0.00	0.00	0.00	0.00	0.07	0.00
Cedar Waxwing	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Chipping Sparrow	0.00	0.00	0.17	0.00	0.00	0.00	0.00
Commom Yellowthroat	0.41	0.25	0.50	0.00	0.00	0.00	0.00
Eastern Bluebird	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Field Sparrow	0.50	0.00	1.00	0.00	0.00	0.00	0.00
Golden-winged Warbler	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gray Catbird	0.00	0.00	0.17	0.00	0.00	0.00	0.00
Indigo Bunting	0.95	0.38	0.83	0.20	0.00	0.00	0.13
Mourning Dove	0.00	0.25	0.00	0.00	0.00	0.00	0.00
Northern Bobwhite	0.05	0.00	0.00	0.00	0.00	0.00	0.00
Northern Cardinal	0.00	0.00	0.50	0.10	0.00	0.00	0.00
Orchard Oriole	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Prairie Warbler	0.14	0.00	0.67	0.00	0.00	0.00	0.00

Appendix 3. Continued.

	Treatment						
	Grasslands		Shrub/pole	Fragmented Forest		Intact Forest	
Species/Guild	Hobet	Daltex	Hobet	Hobet	Daltex	Hobet	Daltex
Song Sparrow	0.09	0.50	0.00	0.00	0.25	0.00	0.00
White-eyed Vireo	0.09	0.00	0.33	0.00	0.00	0.00	0.00
Yellow Warbler	0.36	0.13	0.33	0.00	0.00	0.00	0.00
Yellow-breasted Chat	0.32	0.00	0.67	0.00	0.00	0.00	0.00
Grassland Species							
Bobolink	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dickcissel	0.00	0.75	0.00	0.00	0.00	0.00	0.00
Eastern Meadowlark	0.59	0.75	0.00	0.00	0.00	0.00	0.00
Grasshopper Sparrow	2.27	2.13	0.33	0.00	0.00	0.00	0.00
Henslow's Sparrow	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Horned Lark	0.41	0.13	0.00	0.00	0.00	0.00	0.00
Red-winged Blackbird	1.23	1.75	0.00	0.00	0.00	0.00	0.00
Vesper Sparrow	0.05	0.13	0.00	0.00	0.00	0.00	0.00
Willow Flycatcher	0.18	0.00	0.00	0.00	0.00	0.00	0.00
Habitat Guilds							
Grassland	4.09	5.00	0.33	0.00	0.00	0.00	0.00
Edge	2.86	1.25	6.67	0.35	0.25	0.14	0.13
Interior-edge	0.05	0.00	1.50	2.95	3.75	2.39	3.25
Forest Interior	0.09	0.50	1.00	2.80	2.00	3.93	5.00
Nesting Guilds							
Ground	3.45	4.00	2.50	1.55	1.00	1.82	2.50
Shrub	3.50	2.63	5.50	0.45	0.25	0.29	1.00
Subcanopy	0.00	0.00	1.67	3.10	2.50	2.86	3.75
Canopy	0.05	0.00	0.00	0.80	0.75	0.96	0.75
Cavity	0.00	0.00	0.00	0.75	1.50	0.93	1.00
Total	8.32	7.38	12.17	7.75	6.75	8.00	10.38
Richness	5.50	4.25	9.17	6.70	6.75	6.57	8.63

Appendix 4. Mean abundance of songbird species and guilds in grassland, shrub/pole, fragmented forest, and intact forest treatments on the Hobet, Daltex, and Cannelton mines in 2000.

	Treatment										
	Grasslands			Shrub/pole		Fragmented Forest			Intact Forest		
Species	Hobet	Daltex	Cannelton	Hobet	Cannelton	Hobet	Daltex	Cannelton	Hobet	Daltex	Cannelton
Forest Interior Species											
Acadian Flycatcher	0.00	0.00	0.00	0.06	0.00	0.90	0.50	1.00	1.40	1.12	1.50
Black-throated Green Warbler	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.50	0.10	0.24	0.20
Blue-headed Vireo	0.00	0.00	0.00	0.00	0.00	0.05	0.17	0.50	0.30	0.24	0.70
Cerulean Warbler	0.00	0.00	0.00	0.06	0.00	0.35	0.17	0.30	0.35	0.24	0.60
Eastern Wood-pewee	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00
Great Crested Flycatcher	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00
Kentucky Warbler	0.00	0.00	0.00	0.00	0.00	0.45	0.00	0.00	0.40	0.24	0.00
Louisiana Waterthrush	0.00	0.00	0.00	0.00	0.00	0.25	0.17	0.10	0.05	0.12	0.00
Ovenbird	0.00	0.00	0.00	0.06	0.00	0.75	0.17	0.60	1.25	1.35	1.50
Pileated Woodpecker	0.00	0.00	0.10	0.00	0.00	0.05	0.17	0.10	0.10	0.06	0.00
Scarlet Tanager	0.00	0.00	0.00	0.12	0.06	0.45	0.00	0.20	0.70	0.53	0.90
Summer Tanager	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.15	0.06	0.20
Swainson's Warbler	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.00
Wood Thrush	0.00	0.00	0.00	0.00	0.00	0.55	0.17	0.10	0.70	0.41	0.90
Worm-eating Warbler	0.00	0.00	0.00	0.00	0.00	0.25	0.17	0.10	0.15	0.12	0.30
Yellow-throated Warbler	0.00	0.00	0.00	0.00	0.00	0.20	0.33	0.00	0.10	0.00	0.20
Interior-edge Species											
American Redstart	0.00	0.00	0.00	0.12	0.00	0.20	0.33	0.30	0.85	0.65	0.80
American Robin	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.10
Black-and-white Warbler	0.00	0.00	0.00	0.06	0.00	0.25	0.33	0.30	0.35	0.29	0.40
Black-capped Chickadee	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10
Blue-gray Gnatcatcher	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.06	0.00
Carolina Chickadee	0.06	0.00	0.00	0.12	0.44	0.35	1.00	0.20	0.25	0.18	0.50
Carolina Wren	0.00	0.00	0.00	0.06	0.00	0.25	0.17	0.10	0.10	0.06	0.00
Dark-eyed Junco	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Downy Woodpecker	0.00	0.00	0.00	0.18	0.19	0.20	0.33	0.40	0.00	0.00	0.00
Eastern Phoebe	0.00	0.00	0.00	0.12	0.19	0.00	0.00	0.00	0.00	0.06	0.10
Eastern Towhee	0.17	0.00	0.00	0.53	1.00	0.00	0.00	0.00	0.05	0.00	0.00

Appendix 4. Continued.

Species	Grasslands Hobet	Grasslands Daltex	Grasslands Cannelton	Shrub/pole Hobet	Shrub/pole Cannelton	Fragmented Foresr Hobet	Fragmented Foresr Daltex	Fragmented Foresr Cannelton	Intact Forest Hobet	Intact Forest Daltex	Intact Forest Cannelton
Hairy Woodpecker	0.00	0.00	0.00	0.12	0.06	0.05	0.17	0.00	0.15	0.06	0.00
Hooded Warbler	0.00	0.00	0.00	0.00	0.06	0.10	0.00	0.30	0.60	0.53	0.60
Northern Flicker	0.00	0.00	0.10	0.06	0.06	0.00	0.00	0.00	0.00	0.00	0.10
Northern Parula	0.00	0.00	0.00	0.00	0.06	0.35	0.00	0.60	0.05	0.06	0.30
Red-bellied Woodpecker	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.10	0.12	0.00
Red-eyed Vireo	0.06	0.00	0.00	0.24	0.63	1.70	1.67	1.80	1.40	1.24	1.60
Ruby-throated Hummingbird	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tufted Titmouse	0.00	0.00	0.00	0.12	0.06	0.30	0.33	0.20	0.20	0.35	0.10
White-breasted Nuthatch	0.00	0.00	0.00	0.00	0.06	0.25	0.00	0.20	0.20	0.18	0.00
Yellow-billed Cuckoo	0.06	0.00	0.00	0.00	0.13	0.25	0.00	0.00	0.00	0.00	0.00
Yellow-throated Vireo	0.00	0.00	0.00	0.00	0.00	0.20	0.33	0.20	0.10	0.12	0.10
Edge Species											
American Crow	0.00	0.00	0.00	0.06	0.12	0.00	0.00	0.00	0.00	0.00	0.00
American Goldfinch	0.28	0.25	0.20	0.53	0.56	0.10	0.50	0.00	0.05	0.00	0.00
Baltimore Oriole	0.00	0.00	0.00	0.00	0.55	0.00	0.00	0.00	0.00	0.00	0.00
Blue Grosbeak	0.06	0.33	0.10	0.06	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Blue Jay	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.10	0.12	0.10
Blue-winged Warbler	0.00	0.00	0.00	0.53	0.44	0.00	0.00	0.00	0.00	0.00	0.00
Brown Thrasher	0.17	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00
Brown-headed Cowbird	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.25	0.12	0.00
Cedar Waxwing	0.28	0.00	0.00	0.18	0.50	0.00	0.00	0.00	0.00	0.00	0.00
Chipping Sparrow	0.00	0.00	0.00	0.24	0.31	0.00	0.00	0.00	0.00	0.00	0.00
Common Yellowthroat	0.22	0.17	0.00	0.88	0.69	0.00	0.00	0.00	0.00	0.00	0.00
Eastern Bluebird	0.00	0.08	0.00	0.06	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Field Sparrow	1.06	0.33	0.40	1.35	1.19	0.00	0.00	0.00	0.00	0.00	0.00
Golden-winged Warbler	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Gray Catbird	0.00	0.00	0.00	0.18	0.13	0.00	0.00	0.00	0.00	0.00	0.00
Indigo Bunting	1.00	0.83	1.10	1.47	1.94	0.15	0.50	0.10	0.00	0.18	0.00
Mourning Dove	0.11	0.08	0.00	0.12	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Northern Bobwhite	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Northern Cardinal	0.06	0.00	0.00	0.12	0.38	0.05	0.67	0.10	0.00	0.12	0.00
Orchard Oriole	0.06	0.00	0.10	0.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix 4. Continued.

	Treatment										
	Grasslands			Shrub/pole		Fragmented Foresr			Intact Forest		
Species	Hobet	Daltex	Cannelton	Hobet	Cannelton	Hobet	Daltex	Cannelton	Hobet	Daltex	Cannelton
Prairie Warbler	0.39	0.00	0.20	1.06	1.25	0.00	0.00	0.00	0.00	0.00	0.00
Song Sparrow	0.11	0.58	0.00	0.00	0.19	0.05	0.00	0.00	0.00	0.00	0.00
White-eyed Vireo	0.17	0.00	0.00	0.41	0.50	0.00	0.17	0.00	0.00	0.00	0.00
Yellow Warbler	0.17	0.00	0.00	0.53	0.00	0.00	0.17	0.00	0.00	0.00	0.00
Yellow-breasted Chat	0.28	0.08	0.00	1.24	1.44	0.10	0.00	0.00	0.00	0.00	0.00
Grassland Species											
Bobolink	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dickcissel	0.00	0.33	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eastern Meadowlark	0.39	0.75	0.70	0.06	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Grasshopper Sparrow	3.11	2.67	3.00	0.35	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Henslow's Sparrow	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Horned Lark	0.17	0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Red-winged Blackbird	0.56	1.33	0.30	0.65	0.06	0.05	0.00	0.00	0.00	0.00	0.00
Willow Flycatcher	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Habitat Guilds											
Grassland	3.78	4.50	4.20	1.00	0.31	0.05	0.00	0.00	0.00	0.00	0.00
Edge	3.67	2.17	1.90	6.24	6.69	0.45	1.33	0.10	0.25	0.29	0.10
Interior-edge	0.56	0.17	0.10	1.88	3.06	3.45	3.50	3.00	3.00	2.29	3.10
Forest Interior	0.00	0.00	0.10	0.53	0.19	3.60	2.17	3.50	5.80	4.82	7.00
Nesting Guilds											
Ground	3.61	3.83	3.90	2.29	2.25	1.85	0.83	1.00	2.20	1.94	2.20
Shrub	4.06	3.17	2.10	6.24	6.31	0.55	1.33	0.30	0.60	0.71	0.60
Subcanopy	0.22	0.00	0.10	0.76	1.13	2.25	2.83	2.50	3.05	2.35	3.80
Canopy	0.00	0.00	0.00	0.18	0.13	2.15	1.67	2.50	2.80	1.94	3.50
Cavity	0.06	0.08	0.20	0.65	0.88	1.20	1.67	0.90	0.95	0.88	0.70
	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total	9.17	8.33	6.60	12.18	12.88	9.60	9.17	8.40	10.85	9.18	11.90
Richness	6.00	5.00	3.50	9.00	9.75	8.05	7.00	6.90	8.15	7.24	8.60

Appendix 5. Mean abundance of raptor species for each treatment (GR=grassland; SH=shrub/pole; FR=fragmented forest; IN=intact forest) on each of the 3 mines.

	Cannelton				Daltex			Hobet			
Species	GR	SH	FR	IN	GR	FR	IN	GR	SH	FR	IN
Overall Abundance	0.75	0.48	0.08	0.17	0.75	0.18	0.05	0.83	0.18	0.28	0.33
American Kestrel	0.03	0.00	0.00	0.00	0.15	0.00	0.00	0.03	0.00	0.00	0.00
Peregrine Falcon	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cooper's Hawk	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Accipiter spp.[a]	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Northern Harrier	0.10	0.02	0.00	0.00	0.10	0.00	0.00	0.05	0.00	0.00	0.00
Red-tailed Hawk	0.08	0.02	0.03	0.00	0.05	0.05	0.03	0.05	0.02	0.13	0.08
Red-shouldered Hawk	0.00	0.00	0.05	0.17	0.00	0.03	0.00	0.00	0.05	0.03	0.10
Eastern Screech Owl	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.02	0.00	0.00
Barred Owl	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03
Turkey Vulture	0.50	0.43	0.00	0.00	0.45	0.03	0.00	0.70	0.10	0.13	0.13
Unknown	0.00	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.00	0.00	0.00

[a]Either Sharp-shinned Hawk or Cooper's Hawk.

Appendix 6. Small mammal richness and abundance on each mine in grassland, shrub/pole, fragmented forest and intact forest treatments.

	Mine										
	Cannelton				Daltex			Hobet			
	GR[a]	SH	FR	IN	GR	FR	IN	GR	SH	FR	IN
Species Richness											
1999	-	-	-	-	2.0	1.8	2.5	1.6	-	1.8	2.2
2000	1.0	2.0	1.8	2.0	1.8	1.3	1.0	1.5	1.3	1.5	1.3
Relative Abundance											
Total											
1999	-	-	-	-	18.0	11.3	22.0	15.6	-	13.3	12.0
2000	33.0	25.1	12.1	22.7	8.9	6.2	4.1	22.3	18.2	6.0	2.9
Peromyscus species											
1999	-	-	-	-	13.1	10.0	19.4	14.1	-	11.1	8.7
2000	33.0	21.5	8.0	20.0	4.1	5.6	4.1	21.5	17.6	5.5	2.9
House mouse											
1999	-	-	-	-	4.9	0.0	0.0	0.9	0.0	0.0	0.0
2000	0.0	0.0	0.0	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0
Woodland jumping mouse											
1999	-	-	-	-	0.0	0.0	0.0	0.0	-	0.9	0.0
2000	0.0	0.0	4.1	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Meadow vole											
1999	-	-	-	-	0.0	0.0	0.0	0.1	-	0.0	0.0
2000	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Short-tailed shrew											
1999	-	-	-	-	0.0	1.0	1.9	0.4	-	0.9	2.1
2000	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Eastern chipmunk											
1999	-	-	-	-	0.0	0.3	0.0	0.0	-	0.9	1.2
2000	0.0	0.0	0.0	0.4	0.0	0.3	0.0	0.1	0.0	0.0	1.3

Appendix 6. Cont.

	Mine										
	Cannelton				Daltex			Hobet			
	GR[a]	SH	FR	IN	GR	FR	IN	GR	SH	FR	IN
Eastern woodrat											
1999	-	-	-	-	0.0	0.0	0.0	0.0	-	0.0	0.0
2000	0.0	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Southern bog lemming											
1999	-	-	-	-	0.0	0.0	0.0	0.0	-	0.0	0.0
2000	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.0
Masked shrew											
1999	-	-	-	-	0.0	0.0	0.4	0.0	-	0.1	0.0
2000	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Virginia opossum											
1999	-	-	-	-	0.0	0.0	0.0	0.0	-	0.4	0.0
2000	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Eastern cottontail											
1999	-	-	-	-	0.0	0.0	0.0	0.1	-	0.0	0.0
2000	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0

[a] GR=grassland; SH=shrub/pole; FR=fragmented forest; IN=intact forest.